Appendix E. Potential Effects of Timber Management on Covered Species and their Habitats

CONTENTS

Figures

Tables

E.1 INTRODUCTION

The effects of timber harvest on aquatic life depend on many factors and studies often produce contradictory results (Spence et. al. 1996). Factors that may influence responses include: aquatic species' diversity and adaptability, physical and vegetative conditions and harvest methods, biotic interactions and wide-ranging migratory behaviors can act to reduce impacts of habitat alterations, independent impacts that can accumulate, or interact collectively resulting in compensatory or synergistic responses, and large natural (catastrophic) events that create variable baseline conditions confusing other smaller scale variability.

Despite the difficulties of separating timber harvesting effects from natural disturbance regimes, there has been considerable research on the potential impacts of timber harvesting on aquatic species and their habitats. For example, Chamberlain et al. (1991) summarized four timber harvesting effects that may modify the hydrologic and geomorphic processes and channel formations that determine salmonid habitat:

- Possible increases in peak flows or occurrences of channel-forming flows from increased snow-melt or run-off, resulting in increased bed scour and bank erosion;

- Significant increases in sediment supplies from surface erosion, mass soil movement and bank erosion, leading to channel aggradation, loss of pool volume and degradation of spawning gravels;

- Destabilization of streambanks due to removal of riparian vegetation, physical breakdown, or channel aggradation, increasing sediment supplies and leading to losses of channel formations that constrict flows and promote a diversity of habitat types required by salmonids; and

- Loss or reduction of LWD by direct removal, debris torrents, or management practices that convert riparian corridors to younger stands of predominately hardwoods, contributing to reduced sediment storage sites, reduced pool numbers and volumes, and less rearing habitat for juvenile salmonids.

There has been less research on the potential impacts of timber harvesting on the covered amphibian species, but most of the potential impacts to salmonids and their habitat are believed to also impact the cool-water adapted stream amphibians. In general, timber harvesting activities have the potential to impact all of the Covered Species through altering one or all of the following processes: hydrologic cycle, sediment inputs and transport, large woody debris (LWD) recruitment and distribution, thermal regimes and nutrient inputs.

E.2 ALTERED HYDROLOGIC CYCLES

E.2.1 Potential Effects of Timber Harvesting Activities on Aquatic Habitats

The basic components of the hydrologic cycle are precipitation, infiltration, evaporation, transpiration, storage and runoff. In the Pacific Northwest where annual precipitation is highly seasonal the timing, quantity and quality of rain and snow fall has great influence on salmonid life histories. Thus the interactions of timber harvest activities on the hydrologic cycle are important. This section reviews how timber management activities may influence the hydrologic cycle and the possible impacts on salmonid populations.

Timber harvest temporarily reduces or eliminates leaves and stems. The surface area of this vegetation normally intercepts precipitation for short-term storage that is either evaporated or released as drip. The loss of forest vegetation also reduces the amount of water extracted from the soil by root systems via evapotranspiration and increases soil moisture and pieziometric head. This was demonstrated by Keppeler and Brown (1998) after harvest of second growth redwood forest. Such increases in soil moisture can contribute to increased risk of mass wasting (Sidle et al. 1985, Fig. 10; Schmidt et al. in press). This is discussed further in Section 5.3.2.2. The effect of any reduction in evapotranspiration is typically short lived (3-5 years), as rapid regrowth of vegetation may consume more water than pre-timber harvest amounts (Harr 1977). This is likely to be true in redwood forests as well, in part owing to the stump-sprouting habit of redwood.

The primary effects of timber harvest on surface water hydrology pertain to (Spence et al. 1996):

- peak flows,
- low (base) flows,
- water yield, and
- run-off timing.

Paired watershed experiments to measure changes in flow following timber harvest have been conducted north of the project area (Oregon) and south of the project area (Mendocino County, California). In relatively small watersheds (about 150 to 1200 ac), peak flow magnitude following harvest tends to increase, with the largest increases occurring in smaller runoff events (less than 1-yr) (Beschta et al. 2000; Ziemer 1998). For 1-yr recurrence interval events, peak flow magnitude increased 13-16%; these increases were 6-9% for 5-yr recurrence interval events (Beschta et al. 2000). At Caspar Creek in Mendocino County, increases in peak flow magnitude were about 10% for 2-yr storm recurrence interval events. The effect of timber harvest on peak flows generally diminishes with increasing watershed size and with increasing flow magnitude (Beschta et al. 2000; Ziemer 1998). Effects for larger watersheds are difficult to asses because they are influenced by many additional factors, including regulatory controls on the proportion of the landscape that can be harvested at any given time (e.g., clearcut adjacency and rotation age restrictions adopted by the Board of Forestry) and the extreme variability introduced when attempting to study large basins that experience relatively infrequent major hydrologic events.

The extent of harvest-related changes in hydrology within a watershed may be affected by whether the system is rain or snow dominated. Keppeler and Ziemer (1990, as cited by Spence et al. 1996) found increased summer flows in a Northern California stream following timber harvest but this diminished after five years. In many cases, for rain-dominated systems in the Coast Range, increases in peak flows (particularly in the fall) following timber harvest, are documented (Spence et al. 1996). The principal increases in peak flows following timber harvest in rain-dominated systems are likely as a result of reduced interception and evapotranspiration rates resulting from the loss of vegetation and the more rapid routing of water to stream channels because of soil compaction and roads (Spence et al. 1996; Ziemer 1998). In contrast, generally in snow-dominated systems in the Northwest, peak flows have been shown to change little following timber harvest. In transient-snow systems studies have been somewhat inconclusive as to the effects of timber harvest on peak flows. However, Harr (1986 as cited by Spence et al. 1996) found that in transient-snow systems where harvest had resulted in increased peak flows, the removal of vegetation increased the delivery of water to the soil from the snow-pack during rain-on-snow events. Other research has shown that increased snow melt rates and delivery of water to the soil occurs during rain-on-snow events accompanied by relatively high temperatures and wind speeds (Coffin and Harr 1992, as cited by Spence et al. 1996). The commercial timberlands within the 11 HPAs are entirely rain-dominated. Therefore, the effects of snow-dominated and rain-on-snow hydrology are not an issue for this Plan.

When logging activities compact or disturb surface soils the infiltration capacity is reduced, possibly increasing surface runoff, peak stream flows and sediment inputs. The soil structure of forested hillslopes regulates the downslope movement of water through the soils and into watersheds. On forested hillslopes the infiltration capacity of the soils usually exceeds rainfall or snowmelt intensities so that all water is absorbed by these soils and transported to stream channels through subsurface pathways (Dryness 1969; Harr 1977). Timber harvest activities that compact or disturb the soil can reduce the infiltration capacity of soils and alter the process of subsurface water movement.

Water and sediment from roads can enter stream channels by many mechanisms (Furniss et. al. 2000):

- Inboard ditches that deliver road drainage to stream channels at truck road stream crossings,

- Inboard ditches that deliver flow to culverts, road drainage dips or water bars with sufficient discharge to create a gully or generate a sediment plume that extends to a stream channel,

- Improperly spaced or located road drainage structures that discharge sufficient water to create a gully or generate a sediment plume that extends to a stream channel, and

- Roads located close enough to a stream that fill slope erosion or fill failures result in sediment discharge in to stream channel.

Some studies have shown that forest roads increase peak flows and sediment inputs to small watersheds when as little as 2.5%-3.9% of the watershed is composed of road surfaces (Harr et al. 1975; Cederholm et al. 1980; King and Tennyson 1984). Studies reporting increases in water yield from logged watersheds indicated that these increases were most evident in the start of the fall/winter wet season when rain quickly filled soil pore spaces in the logged areas and then ran off as surface flow. Differences were less apparent later in the rainy season as soil under mature canopies also became saturated, and runoff from harvested and un-harvested areas became similar (Hibbert 1967; Harr et al. 1979). Other studies have also shown that road construction and some timber harvest activities may lead to increased flows in the first (fall/early winter) small rain events but have no significant effect on larger flow events (Wright et al. 1990; Johnson and Beschta 1980).

Many paired watershed studies have found increases in summer base flow and total water yield (Bosch and Hewlett 1982), particularly in humid coniferous forest types. Studies north of the Plan Area in southwest Oregon (Harr et al. 1979) and south of the Plan Area at Caspar Creek in Mendocino County (Keppeler 1998) found increases in both total water yield and seasonal base flows.

Coastal watersheds of northern California receive a majority of their precipitation as rain. However, some watersheds in the Plan Area have upper sections within the transition zone between rain and snow. Along these hillslopes the forest canopy intercepts snowfall, redistributes the snow, shades the snowpack and acts as a windbreak. In these transient areas the snow is generally wet and sticks to the forest canopy longer than colder, drier snow. In transitional areas snow usually reaches the ground in clumps under trees or as snow melt so that snow pack in forested areas tends to vary in distribution and depth compared to logged hillslopes (Berris and Harr 1987).

Snow melt from hillslopes in coastal watersheds is usually the result of warmer rainfall or latent heat in air moisture rather than from solar radiation. Snow packs in transitional areas may accumulate and melt several times during the wet season. When the forest canopy has been removed more of the snow pack is directly exposed to rainfall, warm air and direct sunlight. Harr (1986) reported there was more heat available to melt snow in a clear-cut stand than in an old-growth Douglas-fir stand during a rain storm with a two year recurrence interval. Plot studies in paired watersheds (logged and unlogged) have reported increases in peak streamflow after rain on snow events in the logged areas (Harr and McCorison 1979; Christner and Harr 1982).

E.2.2 Potential Effects on the Covered Species

The effects of temporary changes in watershed yield, peak flow magnitude and timing, and summer base flows on salmonids and key salmonid habitat characteristics are difficult to assess. The life-cycles of salmonids species have adapted to temporal variations in flow conditions by timing the phases of their life cycles to take advantage of seasonal discharges characteristics (Sullivan et. al. 1987). Increased runoff in the early part of the rainy season may, in some cases, benefit salmonids by reducing water temperatures, improving water quality, and providing more flow for immigrating adult spawners. However, a harvest-related increase in peak flows may increase the number of times that channel substrates are mobilized by storm events and potentially damage developing eggs and alevins in redds (Hicks et al. 1991 as cited by Spence et al. 1996). Damage to developing eggs and alevins in redds would constitute take. Channel forming flows may occur more frequently as a result of an increase in peaks flows and thus habitats for spawning, rearing and foraging may be affected, either adversely or beneficially. Increased peak flows may also affect the survival of over-wintering juvenile salmonids by displacing them out of preferred habitats. Displacement of juveniles could cause take if the displacement impairs individual sheltering needs to the extent of killing or injuring individuals. These flow increases could have marginal beneficial effects by increasing available aquatic habitat. Short-term increases in summer baseflows may improve survival of juveniles (Hicks et al. 1991 as cited by Spence et al. 1996) and increase the amount of aquatic habitat. However, these effects are proportional to harvested area and diminish with regrowth of forest vegetation, so the effects are greatest for small watersheds.

The specific effects of altered hydrology on the amphibian Covered Species and their habitat are not known currently and are equally difficult to assess. Green Diamond is not aware of any studies that have addressed this potential effect on species such as the torrent salamander or tailed frog. The speculation is that, in general, these headwater species would be less likely to be affected relative to salmonid species that spawn and rear lower in the watershed. Tailed frog habitat overlaps with the upper reaches of salmonid habitat, and it is possible that increases in peak flow during winter may have a negative impact on larval tailed frogs. This could occur through entrainment of the substrate, which may displace or directly harm the larvae. Further in extreme circumstances, such increases in peak flow could cause take, which may result in local declines in tailed frog populations. However, this would not likely result in long-term changes in the habitat for the species, and therefore it would not likely to result in major changes in populations of the species. Increases in summer low flows due to harvesting activities may be beneficial to larval tailed frog populations, especially during drought years, so it is not possible to know if the overall impact of altered hydrology on tailed frog populations is positive or negative.

Southern torrent salamanders live in seeps and springs and the uppermost reaches of watercourses. The speculation is that increases in peak flow would be unlikely to have any negative impact on this species. Limited field observations of torrent salamanders during high flows suggest that they simply move to the margins of the channel and would not be impacted by entrainment of the substrate. Since torrent salamanders live in aquatic sites with minimal flows, it seems likely that increases in summer low flows would be beneficial for this species. However, they live in association with Pacific giant salamanders that have the potential to prey on or compete with torrent salamanders. Torrent salamanders specialize in utilizing sites with the most minimal flows, so biotic interactions may change with increases in summer low flows. All of these considerations are highly speculative, and Green Diamond does not believe it is possible to predict whether or not altered hydrology would have an impact, positive or negative, on southern torrent salamanders.

Increased runoff and peak flows and decreased infiltration capacity of soils due to timber management and road construction are also correlated with increased sediment inputs to watercourses (Harr et al. 1975; Cederholm et al. 1980; King and Tennyson 1984). The negative effects of increased sediment inputs on the Covered Species and their habitats are described below in Section E.3.

To summarize, the extent to which watershed hydrology is altered by timber harvesting activities and, similarly, the extent to which such altered hydrology may negatively impact the Covered Species, is a function of the amount and timing of those activities in a sub-basin or watershed. Given the cumulative relationship among those activities and this type of environmental effect, it is difficult to assess the potential for these activities to cause altered hydrology itself, and it is also difficult, in turn, to evaluate the potential for altered hydrology to cause take of the Covered Species. For example, as noted above, management-altered hydrology has the potential to harm both the early stages of development (eggs and alevins) as well as over-wintering juvenile salmonids. On the other hand, the effects of altered hydrology may be beneficial for adults returning to spawn in the fall and summer juvenile populations. Therefore, depending on which potentially limiting factors are actually limiting for salmonid production in a given sub-basin, some levels of altered hydrology may be beneficial. However, if other factors are limiting, altered hydrology may cause take and lead to local declines in populations of salmonids. For instance, if summer water temperatures are limiting, increases in summer base flows could be beneficial. In contrast, increases in winter peak flows could cause take and lead to local declines if spawning or over-wintering survival rates were limiting. In conclusion, the potential impacts of altered hydrology are highly complex, and although it has the potential to cause take that could lead to local declines in populations of the Covered Species, the actual impact of various levels of altered hydrology remain unknown. In any event, as a means of avoiding or minimizing and mitigating any negative impacts that could result from altered hydrology, the Plan provides measures to minimize the potential for harvest operations to cause altered hydrology.

E.3 ALTERED SEDIMENT INPUTS AND TRANSPORT

Timber harvest and the construction and use of the associated road system have the potential to increase sediment inputs. Increased sediment inputs from such activities can reduce the quality of aquatic habitats for all six Covered Species through reduced depth of deep water habitats (primarily pools), increased embeddedness of gravel and cobble substrates, and the effects of chronic turbidity on the Covered Species.

Hillslope erosion, sediment delivery to streams, and sediment transport and sorting within streams are natural dynamic processes that are responsible for creating aquatic habitat for the Covered Species. Steep, geologically young, coastal mountains are especially prone to high natural rates of erosion and the Covered Species have evolved in this environment. However, excessive inputs of sediment from a combination of anthropogenic and natural sources can overload a stream’s ability to store and transport sediment, reducing the quality and quantity of aquatic habitat for the Covered Species.

E.3.1 Northern California Sediment Yields and Sources

The variations in bedrock geology, tectonics, and associated geomorphic characteristics in northern California result in different erosion and sedimentation conditions in different stream reaches (the geology and geomorphology of the Plan Area and 11 HPAs are described in the Geologic and Geomorphic Setting Section of the EIS). Sediment production (erosion) may be highly variable depending on the presence or absence of Franciscan mélange and other geologic formations that contain abundant deep landslides and earthflows and locally extensive shearing and faulting in sedimentary rocks. Lisle (1990) cited previous studies discussing factors affecting annual sediment yield in the Eel River, where geologic conditions are most similar to those found on Green Diamond lands south of Redwood Creek. Of note are observations that sediment yield increases with annual rainfall and with drainage area, unlike many other regions. The increase in sediment yield within the drainage area is attributed to abundant deep-seated landslides adjacent to large mainstem river channels, which greatly increase sediment inputs per unit watershed area to the channel network relative to more stable terrain in smaller watersheds. The following data, as well as other data in Table E-1, illustrate this point.

Brown and Ritter (1971) reported mean annual suspended sediment yield for the Eel River to be 1,720 t/km^2/yr (about 4,900 t/mi^2/yr) for a drainage area of 9,400 km^2 (3,600 mi^2). Kelsey (1980) estimated the sediment yield of the upper Van Duzen River to be 2,500 t/km^2/yr (about 7,100 t/mi^2/yr) for a drainage area of 1,500 km^2 (580 mi^2). Both of these watersheds have abundant deep-seated landslides. In comparison, a stream draining an earthflow in mélange terrain was estimated to produce 24,000 t/km^2/yr (about 68,000 t/mi^2/yr) for an area of 3.4 km^2 (1.3 mi^2), or about 10 times more sediment yield per unit area than the basin as a whole. In Redwood Creek (Table E-1), active earthflows yielded about 5 times more sediment than the basin average.

It should be noted that although earthflows are a form of deep-seated landslide, earthflows are less common than rockslides that have slower episodic rates of movement, and that this comparison overstates the rate of sediment production by deep-seated landslides as a whole. Although earthflows may be more persistent sediment sources, rock slides may deliver more sediment in short time periods. For example a debris flow associated with the Floodgate Slide in Mendocino County on the Navarro River (Sowma-Bawcom 1996) delivered at least 200,000 metric tons of sediment from a landslide area of 0.04 km^2 (0.16 mi^2). Hence, stream reaches affected by active deep seated landslides may be more likely to exhibit transport-limited conditions, and are likely to have high suspended sediment loads. Furthermore, data indicating increasing sediment yield with increasing drainage area are also consistent with the hypothesis that sediment deposits in stream channels are a significant sediment source during periods of peak runoff when stream channels are fully occupied by flow and surficial bed armor layers are disrupted.

In contrast to regions where active earthflows and rockslides contribute massive amounts of sediment to streams, Lisle (1990) observed that more competent sandstone units of the Franciscan Formation deliver less sediment. In these areas, hillslope geomorphology is characterized by V-shaped valleys with steep hillslopes where debris slides are the primary mass wasting process. This description is similar to that given for “coherent sandstone” in Redwood National Park (Cashman et al. 1995), with the exception that in Redwood National Park, these characteristics occur on inner gorge slopes. In addition, abundant coarse sediment is generated in erosion events, most of which is of a size that can be transported during annual flood events (Lisle 1990). In these areas, under forested conditions, sediment yields are approximately 300 t/km2/yr (about 860 t/mi^2/yr Lisle 1990). Kelsey (1982) suggests typical rates in the upper Van Duzen River headwaters to range from 80 to 540 t/km^2/yr (230 to 1,500 t/mi^2/yr). Short-term measurements for largely unlogged Franciscan sandstone in Redwood National Park range from about 30 to 110 t/km^2/yr (90 to 310 t/mi^2/yr). Long-term measurements at Caspar Creek, long-term estimates for Freshwater Creek, and short-term measurements for Freshwater and Jacoby Creeks (Table 5-1), all of which include significant logging effects except perhaps Jacoby Creek, have sediment yields in a range characteristic of “competent” Franciscan sandstone, despite the prevalence of weaker geologic materials (Wildcat Group) in Freshwater Creek. Hence, where active deep-seated landslides do not contribute a major component of sediment inputs, sediment yields are approximately an order of magnitude (a factor of 10) lower.

Table E-1. Characteristic Northern California Coast Range sediment yield.

Watershed (Source)	Drainage Area-mi2 (km2)	Sediment Yield-t/mi2/yr (t/km2/yr)
Large Rivers		
Eel River (Brown & Ritter 1971)	3,600 (9,400)	4,900 (1,700)
Van Duzen River 1941-1975 (Kelsey 1980)	580 (1,500)	7,100 (2,500)
Redwood Creek	280 (725)	6,300-7,700 (2,200-2,700)
Earthflows and Rockslides		
Active Earthflows, Van Duzen River (Kelsey 1978)	1.3 (3.4)	71,000 (25,000)
Active Earthflows, Redwood Creek, (2 sites, annual average 1978-1982) (Nolan and Janda 1995a)	0.01-0.05 (0.023-0.13)	32,800 (11,500)
Rock Slide, Navarro River (minimum estimate from volume estimate for associated debris flow, Floodgate Slide, Sowma-Bawcom 1996)	0.016 (0.04)	13,000,000 (4,600,000)
Small Rivers, Few Active Deep Landslides		
Freshwater Creek Sediment Budget (1988-1997, Suspended load estimate from sediment input budget, Pacific Lumber Co. Watershed Analysis (WPN 2001))	13 (34)	340-430 (120-150)
Freshwater Creek Gauge Data (Suspended load yield, Redwood Sciences Laboratory, WY 2000)	13 (34)	380 (130)
Jacoby Creek (Suspended load yield extrapolation from data, Lehre and Carver 1985)	14 (36)	440 (155)
Redwood Creek (Suspended load yield extrapolation from data for 1973-74, Nolan and Janda 1995b)	0.6-4.0 (1.6-10.3)	90-310 (30-110)
North Fork Caspar Creek (Suspended load yield, 1990-1995, post-logging period, Lewis 1998)	1.8 (4.7)	130 (47)
North Fork Caspar Creek (Suspended load yield, 1963-1995, Cafferata and Spittler 1998)	1.8 (4.7)	380 (130)

These data suggest that aquatic ecosystems and organisms have evolved with relatively high levels of erosion and sedimentation, and that in watersheds where deep seated landslides and earthflows characteristic of Franciscan mélange are common and active, high levels of erosion and sedimentation are to be expected, regardless of management influences. Furthermore, the data suggest that smaller watersheds where large, active landslide complexes are found, such as those found on the Eel River and Redwood Creek, are less likely to have extremely high erosion rates. Thus, anadromous fish may have historically had access to watersheds with lower erosion and sedimentation rates, despite extreme erosion rates in some locales. These generalizations are limited by local geologic and management history at the watershed scale, however, it is fair to say that regardless of the efficacy of any future efforts to prevent excessive erosion from management, there will be episodes of high erosion and sedimentation rates at various spatial and temporal scales in streams draining the northern California forest landscape.

Comparison of erosion rates attributed to forest management to background erosion rates provides valuable perspective on the significance of high natural erosion rates and management impacts. Recent investigations of northern California erosion rates at the watershed scale have been conducted by a variety of contractors for the U.S. Environmental Protection Agency (EPA) and the North Coast Regional Water Quality Control Board (NCRWQCB). These studies are accessible via the internet at http://www.epa.gov/region09/water/tmdl/index.html. Several of these studies were analyzed to develop a common quantitative format allowing for the results to be compared and to assess whether any general conclusions may be drawn with respect to harvest effects and road effects on erosion rates in the Plan Area. The most general form of the results of this review is presented in Table E-2 below.

Table E-2. Summary results of recent regional erosion source studies in northern California.

Watershed	Background[1] (% of total)	Management Sources	
		Mass Wasting[2] (% of total)	Surface Erosion, Road Erosion, Other Sources[3] (% of total)
Sproul (S.Fk.Eel)	24	19	57
Tom Long (S.Fk.Eel)	71	5	24
Hollow Tree (S.Fk.Eel)	43	24	33
Noyo River	58	13	28
Upper S. Fk. Trinity	66	11	23
Lower S. Fk. Trinity	68	21	10
Hayfork Cr. (S. Trinity)	49	1	50
Freshwater Cr.4	40	16	44
Mean	**52**	**14**	**34**
Range of Values	**24-71**	**1-24**	**10-57**

Notes
1 Includes streamside landslides thought to be of natural origin and all deep seated landslides.
2. Includes road and harvest related slides; harvest related slides are typically assumed to be triggered by harvest if they are observed in harvested area, regardless of actual triggering mechanism.
3 Road surface erosion (sheet and rill erosion of road tread and cut slopes) is the dominant surface erosion process assessed; additional road erosion is from gullies and other road-drainage related erosion. Other sources (e.g. bank erosion) are relatively small.
4 Pacific Lumber Co. Watershed Analysis (WPN 2001)); all others are TMDL studies by USEPA or NCRWQCB.

These data on erosion sources represent conditions over roughly the past 30 years, with the implementation of the California Forest Practice Act beginning early in the period for which the data are summarized. The studies shown were selected in part because they are generally comparable with respect to the erosion processes for which rate estimates were developed and to techniques used to develop erosion rate estimates. The summary presented in Table E-2 should be interpreted with some caution owing to remaining differences in the methods employed and differing scales of different studies. The mean values reported in Table E-2 are in agreement with a similar, prior investigation based on intensive erosion surveys of Redwood National Park (Hagans and Weaver 1987), suggesting that the data in Table E-2 are reasonably well-supported and representative of regional conditions. On the basis of these data, management-related erosion at the watershed scale typically induces increases in erosion of about 100%, ranging from about 30% to over 300%. The data indicate that management erosion sources other than mass wasting, primarily road-related erosion, are believed to be at least as large or larger than management-related mass wasting (Lower S. Fk. Trinity in Table E-2 is the lone exception).

E.3.2 Erosion Sources and Processes

E.3.2.1 Surface Erosion

A common source of sediment input to watersheds is surface erosion. Surface erosion can be major contributor of sediment in areas where soils are composed of granite or highly fractured marine sedimentary rocks (Furniss et al. 1991). Surface erosion is a two-part process in which particles are first detached and then transported downslope. The two hydrologic processes that transport surface erosion are channelized erosion by constricted flows (rilling and gullying) and sheet erosion in which soil movement is non-channelized (rolling and sliding) (Swanston 1991).

Increases in channelized and non-channelized erosion occur when the infiltration capacities of soils are reduced by management activities, large storm events or fires. Chamberlain et al. (1991) reported that the potential for surface erosion is directly related to the amount of bare soil exposed to rainfall and runoff. A study in Redwood National Park using erosion pins (Marron et al. 1995) found that erosion following logging on soils derived from sandstone was not significant to the watershed sediment budget, but that logging on soils derived from schist may be significant. Higher erosion rates tended to occur where rill erosion was more common, which was associated with tractor-harvest, and to a lesser extent, cable yarding, on schist soils. The study examined soil detachment and local ground surface lowering, but did not assess delivery of eroded sediment to streams. Hagans and Weaver (1987) analyzed the data used by Marron et al. (1995), as well as data on percent bare soil following harvest and data on sediment delivery to streams from surface erosion processes on logged areas, including skid trails, for the lower Redwood Creek basin for the period c. 1954-1980, and concluded that only 4% of erosion was caused by sheet and rill erosion. Rice and Datzman (1981) conducted detailed surveys in northern California of 102 harvested plots averaging about 11 acres in size over a range of geologic and slope conditions. In aggregate, they found that two-thirds of the observed erosion was associated with roads, landings or skid trails. Surface erosion in the form of rills and gulleys not associated with roads, landings or skid trails (i.e. harvested areas) accounted for about five percent of total erosion.

Surface erosion by rainsplash and sheetwash processes from roads (including cut slopes), watercourse crossings, landings, skid trails and ditches may all contribute to substantial increases in surface erosion and increased delivery of sediments into stream channels (Reid and Dunne 1984; Luce and Black 1999; Duan 2001). Road erosion estimates in Table E-2 include substantial quantities of sediment from rainsplash and sheetwash processes delivered to streams.

E.3.2.2 Mass Wasting

In steep mountainous terrain, mass soil movement is a major type of hillslope erosion and sediment source in watersheds (Sidle et al. 1985; Swanston 1991). The frequency and magnitude of mass soil movements is governed by hillslope gradient, level of soil saturation, composition of dominant soil and rock types, degree of weathering, type and level of management activities, and occurrence of climatic or geologic events.

Mass soil movements are usually episodic events and tend to contribute significant quantities of sediment and organic debris to stream channels over time intervals ranging from minutes to decades (Swanston 1991). The resultant sediment and organic debris may have a profound effect on a stream channel including large increases in coarse and fine sediments, shifts of existing bed-load, and increases in woody debris that can lead to partial or complete stream blockages.

Forest management practices can affect slope stability by changing vegetative cover, hillslope shape, and water flow above and below the ground surface. Different forest management operations have distinct effects on the factors that control slope stability. For two of the major components of forest management operations—road construction (and to a lesser extent skid trail construction) and harvesting trees—the potential consequences with respect to shallow landslide processes and slope stability are relatively well known. These are described briefly below, with more detailed discussion following.

Road and skid trail construction may:

1. create cut slopes and fill slopes too steep to be stable,

2. result in deposition of sidecast material (spoils) that overburdens and/or oversteepens slopes, and

3. divert and/or concentrate both surface and subsurface runoff.

Harvesting trees:

1. reduces effective soil cohesion by disrupting networks of interlocking roots from living trees, and

2. increases soil moisture by reducing interception of precipitation and evapotranspiration of soil water.

The actual influence of specific forest management activities on slope stability, however, depends on the design and construction of the road network, density of residual trees and under-story vegetation, rate and type of revegetation, topography, material strengths, patterns of surface and subsurface flow, and patterns of water inflow (Sidle et al. 1985; Yoshinori and Osamu 1984). Landslide rates associated with roads are generally much greater than landslide rates associated with timber harvest alone (Sidle et al. 1985).

Changes in canopy interception and evapotranspiration following timber harvest tend to increase soil moisture. This is significant because greater soil moisture reduces the amount of precipitation from a given storm event required to cause soil moisture levels to reach a critical level. This relatively simple qualitative statement regarding soil moisture does not account for complex spatial and temporal effects of vegetation change on hillslope hydrology that could affect slope stability. The potential hydrologic effects are less understood in comparison to the foregoing effects, and therefore have greater attendant uncertainty with respect to effects of forest management.

Timber harvest activities (falling and yarding) not directly associated with roads can increase direct sediment input to streams through surface erosion and mass wasting. Timber harvest may increase the amount of bare soil exposed to rainfall and runoff, leading to increased surface erosion. The occurrence of mass wasting may also increase after timber harvesting, depending in part on the type and intensity of harvest methods (Rood 1984; Swanson et al. 1987). Sidle et al. (1985) reviewed mass wasting surveys concluded during the 1970's and found that mass wasting rates (landslide volume per unit area per unit time) increased from 0 to 40 fold, with the median increase being 3.7 times the rate for undisturbed forest. The substantially lower proportion of increase in erosion from harvest-related landslides relative to data in Table E-2 may be attributable to at least three factors. First, Sidle's review represented historical harvest practices prior to 1980, whereas the reviews in Table E-2 begins in the 1970's. Second, Sidle's review does not distinguish between eroded sediment and delivered sediment, and probably represents erosion rates rather than sediment delivery rates. Consequently, sediment eroded and subsequently deposited on hillslopes is presumably included in Sidle's ratios. Third, the background erosion rates in northern California (e.g. Table E-1) are generally higher than in the areas cited in Sidle's review; hence the proportional increase related to harvest would be lower in the Plan Area (consistent with Table E-2).

Separating the effects of timber harvest activities from the associated yarding, construction, maintenance and use of skid roads and the forest road system may be difficult. Further, the results vary between watersheds. Most studies indicate that the sediment inputs from timber harvesting alone are less than those of the associated road network (e.g. Table E-2, also see Sidle et. al. 1985, Raines and Kelsey 1991, Best et al. 1995).

The Oregon Department of Forestry study of landsliding associated with the high intensity, low frequency storms and flood events in February and November 1996 (Robison et al. 1999) revealed that in areas with slopes > 60%, average sediment delivery was about 2.5 times higher for 0-9 year age class of timber compared 100-year plus age class. In contrast, for the 10-30 year old age class, half of the study areas had lower erosion rates compared to 100-year plus age class. The results reflect the short-term impact of a very large storm and therefore likely overestimate the long-term impact of harvesting.

Federal and state regulatory programs have recently required development of TMDL calculations for designated watersheds in northern California. The primary tool utilized to date for development of TMDLs has been quantitative sediment source assessments (sediment budgets for erosion sources). Although the data collected from the TMDL studies is not sufficient to quantitatively evaluate the impact of harvesting, the data does suggest that harvest-related slides, on average, contribute less sediment than background ("natural") and road-related erosion sources.

In connection with regulatory action by the State of California North Coast Regional Water Quality Control Board against Pacific Lumber Company (PALCO), sediment source studies were conducted by Pacific Watershed Associates for Bear River (PWA 1998b), Jordan Creek (PWA 1999b), North Fork Elk River (PWA 1998a), and Freshwater Creek (PWA 1999a). The latter study included a landslide inventory that was expanded in a subsequent Watershed Analysis. The results from the Bear, Jordan and North Elk can be interpreted to reveal a 2.3 to 11 times increase landslide rates associated with harvesting when the effects of recent harvest and high intensity, low recurrence rainstorms and floods in 1995 and 1997 are considered over a period of about 25 years. However, these interpretations on the impacts of harvesting must be viewed with caution, since the majority of those landslides were associated with very large storms that occurred over a very short time period.

The PALCO Freshwater Creek Watershed analysis is the only recently published analysis, which specifically looked at landslide rates in clearcut, partial cut, and forested areas in northern California. Landslide rates (#/acre/yr) in clearcut areas were on average 2.3 times higher compared to second growth unthinned areas. The impact in headwall swales was higher, about 5 times higher for clearcuts compared to second growth. No difference was apparent between thinned second growth and uncut second growth.

Preliminary landslide data from Hunter Creek mass wasting assessment revealed that landslide delivery rates in clearcut units were between 1.0 and 1.7 times higher than uncut forested areas for the 1958 to 1972 air photo period. The majority of this impact was associated with the intense 1972 storms and long term impacts will likely be less. Cafferatta and Spittler (1998) found little difference between landslide rates in clear cut areas and mature forest in northern California.

These results suggest that landslide rates on harvested areas do not uniformly increase, and that there is considerable uncertainty regarding the circumstances under which reported increases have occurred. It is possible that increased risk avoidance in development of timber harvest plans under present-day professional and regulatory standards help to explain the results of the three studies noted above and those in Table E-2 relative to earlier studies (Sidle et al. 1985).

The changes in physical processes associated with timber harvesting (timber removal alone) are reduced root reinforcement of shallow soils by root-wood deterioration and, to a lesser extent, temporary increases in water input and soil moisture because of reduced evapotranspiration and reduced rainfall interception (or increased throughfall). Whether or not sediments related to timber harvest activities actually enters a watercourse is related to local topography and the proximity of the timber harvest to a watercourse.

E.3.2.2.1 <u>Reduced Root Reinforcement</u>

After forest removal, the gradual decay of small tree roots can predispose certain slopes to failure (Burroughs and Thomas 1977; O'Loughlin and Ziemer 1982; Wu and Swanston 1980; Ziemer 1981a; Ziemer 1981b; Ziemer 1981c; Ziemer and Swanston 1977). Root systems contribute to soil strength by providing effective cohesion (Sidle et al. 1985). Studies have shown that most of the original root reinforcement is lost 4 to 15 years following harvest in a Douglas-fir and pine forest. Redwood and hardwood stands, which dominate the commercial timberlands in the 11 HPAs, resprout after cutting; in these stands a significant loss in root strength is less likely to occur. Landslide susceptibility may also be a function of species composition and spatial variability of root reinforcement (Schmidt et al. in press).

The timing of landsliding, however, may not always be coincident with maximum root deterioration because of the relatively low frequency of occurrence of required storm thresholds (Cafferata and Spittler 1998). Recently harvested areas in the Elk River

(PWA 1998b) and Bear Creek (PWA 1998a) watersheds in Humboldt County experienced unusually high landslide rates in part because a series of low frequency, high intensity storms between 1994 and 1997. These landslide rates may reflect hydrologic influences as much or more than root strength losses. On the other hand, according to Montgomery et al (2000), storms with recurrence intervals less than four years are associated with many landslides in the Oregon Coast Range. In any case, the extent to which losses in root reinforcement of soil trigger landslides depends in part on the intensity of harvest and in part on the timing of subsequent rainstorms, particularly in the “window” of reduced root reinforcement up to about 15 years.

The effect of root strength is most apparent in shallow cohesionless soils on steep slopes (Chatwin et al. 1994; Sidle and Swanston 1982). Soil cohesion from root systems rarely extend to a depth of > 1.5 ft in coastal Oregon (Schmidt et al. in press). Most of the soil reinforcement by roots is therefore a function of the lateral spread of roots. The root strength in the upper portion of the soil column provides little, if any, additional stability to deep-seated landslides where failure planes often exceed 20 feet in depth (Sidle et al. 1985; Yoshinori and Osamu 1984) or in soils that have high cohesion. Landslide susceptibility may also be a function of species composition and spatial variability of root cohesion (i.e. spacing and distribution of root networks of conifers, hardwoods and shrubs; Schmidt et. al. in press).

Modeling studies of shallow landslides and the effects of different silvicultural systems on root strength suggest that partial cutting, thinning and shelterwood techniques result in substantial increases in root strength and substantial decreases in probability of slope failure (Sidle 1992; Krogstad 1995). In addition, understory vegetation often represents a substantial component of total root cohesion (Schmidt et al. in press), suggesting that efforts to suppress understory vegetation following timber harvest may reduce root cohesion and increase the potential for shallow landslides on susceptible slopes.

E.3.2.2.2 *<u>Decrease in Evapotranspiration and Rainfall Interception</u>*

Evapotranspiration can influence soil water recharge and subsurface flow and thus has the potential for affecting shallow and deep-seated slope stability. The removal of vegetation from a hillside may locally increase the level of ground saturation by reducing the amount of water intercepted and transpired by the canopy (Keppeler et al. 1994; Keppeler and Ziemer 1990; Swanson et al. 1987; Swanson 1981). Where slopes are marginally stable, the resulting increased soil moisture and higher pore pressures may increase both the rate and duration of slope movement. The effects of reduced transpiration and rainfall interception are diminished as vegetation becomes re-established.

Most shallow slides are triggered by peak groundwater levels during high-intensity rainfall events in the winter months when vegetative transpiration rates are already low. Once winter moisture conditions are attained, generally by early December, the difference in soil moisture between logged (clear-cut) and unlogged slopes is virtually indistinguishable (Gray 1977). On the other hand, reduced evapotranspiration may allow near-surface soils to become wetter sooner and stay wetter longer and therefore expose the slope to a potential triggering storm event for a longer period during the wet season.

Canopy interception during storms reduces water delivery to the soil by about 15 to 35% in coniferous forests (Dunne and Leopold 1978). To the extent that some landslides are triggered by relatively short duration bursts of high intensity precipitation (Wieczorek 1996), the loss of canopy would be expected to increase the potential for landslides in susceptible areas. Over seasonal time periods, median canopy interception for coniferous forests is about 22% (Dunne and Leopold 1978). This increment of additional water input to soil that could result from timber harvest could increase the frequency of critical soil moisture conditions when landslides are most likely to occur.

The actual effect of an individual timber harvest on porewater pressures, however, is site specific, dependent upon the characteristics of the underlying parent material (hydraulic conductivity, storativity, shear strength, etc), hillslope geometry, water input, and density of the residual stand. Little change in porewater pressures will be realized in materials with high hydraulic conductivity (i.e. drain rapidly) and/or high storativity (i.e. high porosity) compared to materials that have both low hydraulic conductivity and low storativity.

Upslope clearcut harvesting may potentially influence downslope failures by altering the water balance at the hillslope scale. This was suggested as a potential mechanism contributing to observed landsliding in Bear Creek (see PWA 1998a), a tributary to the Eel River in southern Humboldt County, California. The hypothesis holds that the scale dependent effect would be greatest on larger drainages and where the entire slope from ridge top to near valley bottom was harvested (Tom Spittler, pers. comm., 1998). Modeling studies revealed potentially significant effects of upslope harvest on the stability of historically-active, deep-seated landslides at a site in western Washington (Miller and Sias 1998). Additional research would be required to test this hypothesis and to quantify the attributes where such a process is most important.

E.3.2.3 Deep-Seated Landslides

Natural mechanisms that may trigger deep-seated landslides include intense rainfall, earthquake shaking, and erosion of landslide toes by streams. It is generally acknowledged that deep-seated landslides (earthflows and rockslides) may be destabilized by undercutting of the landslide toe (e.g. by streambank erosion or excavation of road cuts), by adding significant mass to the landslide body (e.g. disposing of spoils from grading or excavation projects), or by significantly altering the groundwater conditions in a landslide (e.g. diversion of road drainage into head scarps or lateral scarps) (TRB 1996, Ch. 16). Deep-seated landslides may also be affected by these hydrologic changes associated with reduced evapotranspiration reduced canopy interception during rainstorms (DMG 1997). Potential increases in groundwater associated with timber harvest in areas upslope of active deep-seated slides may also be important.

Reduced evapotranspiration may add substantially to the annual groundwater flux. Measurements of change in annual stream runoff provide an estimate of the magnitude of change in evapotranspiration following timber harvest. Data from two watershed experiments at Caspar Creek in the redwood region of coastal California using partial cut (65% volume removal) and clearcut harvest (50% of watershed) techniques both indicated an average annual increase in runoff of 15% (Keppeler 1998). The increase in groundwater implied by these experiments is a potential risk factor for increased activity of deep-seated landslides.

Miller and Sias (1998) modeled the effect of timber harvest on groundwater conditions and slope stability of a large, deep-seated landslide in glacial lacustrine sediments adjacent to a large river channel. They predicted that timber harvest in the groundwater recharge area of the landslide would produce very small decreases in the factor of safety, suggesting that harvest would contribute to landslide movement only if the landslide were at or near the threshold of stability. This suggests that active deep-seated landslides are most likely to be affected by harvest-induced changes in groundwater, while inactive and dormant slides are less likely to be affected.

Iverson (2000) developed a model of landslide triggering in response to rainfall. For large deep seated slides with low hydraulic conductivity and large contributing drainage area, landslide force balances driven by hydrologic factors change over periods of time on the order of months to years. Consequently, large deep seated landslides are expected to be sensitive to long-term cycles of precipitation. While it is implied that changes in evapotranspiration could also affect landslide force balances, this potential effect would be substantially reduced by limiting the extent or intensity of harvest to avoid sharp, persistent declines in evapotranspiration.

The relatively few regional empirical landslide studies have produced varying conclusions on the effect of timber harvesting on earthflow stability. Short-term increases in ground displacement following clear cutting have been documented on several active earthflows in the Coast Range and Cascades of Oregon (Pyles et al. 1987; Swanson et al. 1988; Swanson et al. 1987; Swanston 1981). In contrast, work by Pyles et al. (1987) on the Lookout Creek earthflow in central Oregon concluded that timber harvesting was unlikely to induce a large increase in movement, primarily because the slide was well drained.

In summary, previous studies suggest that forest management activities can potentially increase the occurrence or rate of movement of deep-seated landslides. Recognition of active landslides and avoidance of management practices that are known to increase risks of movement can reduce the overall risk of erosion associated with deep landslides. Site-specific conditions pertaining to individual slides will always be important in development of site-specific forest management plans, nevertheless, substantial uncertainty is likely to remain regarding predicted effects of management on slide activity. Deep landslides are relatively common, naturally occurring geologic features in northern California that will continue to generate substantial quantities of sediment delivered to streams, regardless of management influences.

E.3.2.4 Sediment Input from Roads

In the past 25 years studies and reports have shown that road construction for timber harvesting causes great increases in erosion rates within a watershed (Haupt 1959; Gibbons and Salo 1973; Beschta 1978; Cederholm et al. 1980; Reid and Dunne 1984; Swanson et al. 1987; Furniss et al. 1991). Roads affect watersheds by modifying natural drainage patterns and by accelerating erosion and sedimentation, thereby altering channel stability and morphology. If proper construction techniques and maintenance practices are not followed, sediment increases following road construction can be severe and long-lasting. Gibbons and Salo (1973) concluded that the sediment contribution per unit area from forest roads is usually greater than that contributed from all other timber harvesting activities combined. Cederholm et al. (1980) reported a significant positive correlation between the percentage of basin area in road surfaces and percentage of fine sediments (less than 0.85 mm) in spawning gravels.

Forest road systems and their associated watercourse crossings in steep coastal watersheds have the potential to be a major cause of mass soil movements (Best et. al. 1995; Sidle et al. 1985; many others). Road inventories conducted in the Pacific Northwest have reported that erosion from older roads may contribute 40 to 70 percent of the total sediment delivered to the system (Best et al. 1995; Durgin et al. 1988; McCashion and Rice 1983; Raines and Kelsey 1991; Rice and Lewis 1991; Swanson and Dryness 1975).

Raines and Kelsey (1991) developed a sediment budget for Grouse Creek, a tributary to the South Fork Trinity River. These authors concluded that within the Grouse Creek watershed, erosion rates from managed lands were 1 to 6 times higher than erosion rates in unmanaged lands, and erosion rates from roads were 20 to 140 times higher, depending on the time period studied. Road related erosion was the largest single source of sediment volume per unit area (Raines and Kelsey 1991). Sidle et al. (1985) reported that mass soil movements associated with forest roads were 30 to 346 times greater per unit area (median=125) compared to undisturbed forest, consistent with findings by Raines and Kelsey (1991) for Grouse Creek. Fluvial erosion of gullies related to road drainage problems (plugged culverts and resulting stream diversions) accounted for 16% of the sediment budget for Garrett Creek in Redwood National Park (Best et al. 1995). Cederholm et al. (1980) reported that in Washington’s Clearwater watershed 60% of road related sediment production was from associated hillslope failures and that road surfaces accounted for 18-26% of all sediment production. These and many other studies demonstrate that roads are typically the dominant element in management-related erosion in forested upland watersheds.

Increases in hillslope failures due to roads are affected by variables such as hillslope gradient, soil type, soil saturation, bedrock type and structure, management levels and road placement. However, the literature suggests that road placement is the single most important factor because it affects how much the other variables will contribute to slope failures (Anderson 1971; Larse 1971; Swanston 1971; Swanston and Swanson 1976; Weaver and Hagans 1994). Specific road-related landslide triggering mechanisms responsible for road-related mass wasting are described below.

Recently, techniques have been developed to improve the construction and maintenance of forest roads which minimize erosion and sedimentation and should be incorporated into new and existing road networks (Weaver and Hagans 1994). However, a road construction and maintenance crew that is skilled in these techniques and motivated to do quality work is vital to the success of a low impact forest road network.

E.3.2.4.1 <u>Oversteepening</u>

Midslope roads may require cut slopes which create a slope angle too steep for stability. These steep new slopes, coupled with the loss of root strength and increased water inputs (as discussed below) may be subject to surface erosion and landsliding. Modern road building techniques make these roads infrequent in new construction.

E.3.2.4.2 *Fill Materials*

Placement of thick unengineered fill onto steep and potentially unstable slopes can lead to slope failures by increasing slope weight and altering local groundwater conditions. In addition, inadequate or poorly designed road drainage can result in runoff diverted onto loose and potentially unstable fill material. Saturation of fill significantly increases potential for slope failure. Further, loss of soil strength from decomposition of organics incorporated within the fill may ultimately result in slope failures several years or decades after road construction.

E.3.2.4.3 *Concentration of Road Drainage*

The concentration of road runoff from inadequately or improperly spaced road drains and/or the augmentation of runoff resulting from rerouting flow in road ditches from one drainage basin to another can saturate the soil more quickly and more frequently, leading to increased likelihood of slope failure. Undersized culverts that become plugged with debris and overtopped during large rainfall events can lead to failure of the fill at the crossing or, if runoff is diverted down the road, failure of an adjacent slope.

Whether or not sediment from road-related surface erosion or mass wasting events actually enters a watercourse is related to local topography, the proximity of the road to a watercourse, and whether or not it the road is connected hydraulically to that watercourse.

E.3.2.4.4 *Reduced Bank Stability*

Timber harvest in riparian areas has the potential to reduce bank stability and reduce the capacity of the riparian zone to act as a filter strip for sediment transport from upslope sources. These potential losses of riparian function result from soil compaction and exposure via heavy equipment operation, and loss of vegetative root strength and structure due to the removal of harvested trees and damage to other riparian vegetation.

Reducing the capacity of the riparian zone to act as a filter strip essentially increases sediment input to watercourses, the effects of which are described above. Loss of bank stability may lead to channel widening, increased sediment input (from the eroding banks), and a decrease in habitat depth and complexity. Channel widening in turn reduces canopy cover, increasing stream temperatures, and reducing organic input to the stream.

E.3.3 Sediment Transport Processes

The following discussion addresses several aspects of erosion and sedimentation processes. First, sediment transport mechanics are described, followed by a discussion of general watershed scale erosion and sedimentation phenomena. The spatial and temporal relationship between specific erosion processes and sediment transport processes are then developed in greater detail, including considerations regarding timing of sedimentation and effects on aquatic habitat. Next, a discussion of natural factors that mitigate sedimentation effects is presented, followed by management considerations regarding strategies available to minimize sedimentation impacts to aquatic habitats.

This Section discusses the distinctive characteristics of three modes of sediment transport in stream channels: bedload, intermittent suspended load, and suspended load. Although each of these processes corresponds to a generally consistent size range of sediment, it should be noted that these processes occur over a physical continuum, and that there is substantial overlap between these modes of sediment transport. Depending on the intensity (i.e. velocity) of stream flow, the sediment transported in one mode may be transported in another mode. Many textbooks provide a description of sediment transport mechanics (e.g. Richards 1982, Raudkivi 1990, Yang 1996).

E.3.3.1 Bedload Sediment

The typical size of material transported primarily as bedload in upland streams is gravel (2 mm to 64 mm diameter) and cobble (64 mm to 256 mm diameter). Larger material (boulders) are also transported as bedload, however, sediment particles of this size move relatively slowly and are more likely to form nodes of stability in stream channels (i.e. boulder steps or transverse bars, Grant 1990).

Bedload is transported by sliding, rolling, or skipping along the streambed. Bedload particles are rarely found in the water column far above the bed. Bedload sediment is typically routed through mountain channel systems slowly, with average annual transport distances from tracer studies of about 300 ft, ranging from about 60 to 1500 ft (NCASI 1999, p. 289). The volume of bedload sediment deposits is typically large in comparison with the annual transport rate.

Bedload sediment is broken and abraded as it collides with other sediment clasts on the bed or in transport; this gradual process of breakage and declining size is known as attrition. The attrition process converts a portion of the bedload to suspended load as larger sediment clasts produce smaller sediment particles. The attrition rate is usually estimated as a function of transport distance in the channel network. The magnitude of attrition varies, but as much as half of bedload material may be converted to suspended sediment over transport distances of about 20 km (Collins and Dunne 1989). Where bedrock is extremely weak (e.g. Wildcat Group rocks near Humboldt Bay), however, the attrition rate may be much higher, and where bedrock is relatively strong, the attrition rate much lower. Intermittent suspended load (also called “saltation load” by Raudkivi (1990)) is typically comprised of fine gravel and coarse sand. It is transported partly in contact with streambed, and partly in suspension, depending on flow intensity and local channel morphology. These sediment sizes are often found in sorted deposits in the lee of channel obstructions or in pools, and are typically finer than typical median grain size on the surface of point bars and alternate bars. Intermittent suspended load is transported through channel systems more quickly, provided it is not deposited underneath coarse armor layers of bed and bar deposits. The typical annual velocity of intermittent suspended load is between that of bedload and suspended load, and is on the order of 1000’s of ft to miles.

E.3.3.2 Suspended Sediment

Sand, silt and clay sizes (< 2 mm diameter) comprise the suspended sediment load in most upland stream systems. The sand fraction (> 0.06 mm and < 2 mm) is often a major constituent of the intermittent suspended load and a substantial constituent of the bedload. In many low-gradient rivers, sand is the dominant component of the bedload.

Such conditions are found at the mouths of several coastal watersheds in northern California.

Suspended load is transported in suspension in the water column in relatively low-intensity flows. It typically is transported through the channel system rapidly; sediment velocity for suspended load is nearly equal to water velocity. If suspended sediment is present in or on the margins of channels it will be entrained rapidly with increasing stream discharge. This suspended sediment can be subsequently deposited in low-velocity areas downstream as stream discharge declines. Sediment of this type is rarely deposited in large quantities within the streambed in upland channel networks except in low-velocity environments such as unusually low gradient or hydraulically rough reaches, channel margins, side channels, and behind flow obstructions.

A finer component of the suspended load is sometimes referred to as “wash load” (Raudkivi 1990; Reid and Dunne 1996). Wash load is usually comprised of clay and fine silt, and is distinctive in that once entrained in the water column of a stream, it will not settle out. Hence, this size fraction is found in only very small quantities in the bed of upland streams.

Much of the suspended load is removed from the upland stream system very rapidly and is deposited in floodplains, estuaries and offshore marine environments. Suspended load accounts for about 70 to 90% or more of the total sediment load in northern California watersheds. This includes the wash load, the suspended load and, depending on measurement technique, some portion of the intermittent suspended load measured

Suspended load transport in many northern California streams (e.g. Caspar Creek, Lewis 1998) is correlated with turbidity (an optical characteristic of water quantifying its clarity or cloudiness). Hence, the supply of suspended load sediment size fractions is the chief control on stream turbidity, a measure of water quality used by the California Regional Water Quality Control Board in its Basin Plan for northern coastal California. The silt and clay fraction in the suspended load (this is typically equivalent to the wash load in most upland streams in northern California) strongly influences turbidity, hence control of sediment sources rich in silt and clay will provide the greatest reduction in turbidity.

E.3.3.3 Watershed-scale Sediment Transport Concepts

The relationship between sediment inputs to a channel network and sediment transport capacity of the channel network will have a strong influence on channel sedimentation status (e.g. Montgomery and Buffington 1993, Buffington and Montgomery 1999). For example, channel systems that are said to be “transport-limited” are expected to contrast with “supply-limited” systems. The influence of sediment supply on bedload transport processes has been the subject of much research, including recent field-based modeling work (Lisle et al. 2000) that suggests that a higher proportion of the streambed tends to be mobilized by competent flows in channels with a higher sediment supply. In addition, some studies suggest that high sand loads in a gravel bed stream may increase bed mobility (Iseya and Ikeda 1987). Increased bed mobility would increase bed scour potential.

E.3.3.3.1 Transport-limited Channels

Transport-limited channels are defined by high sediment supply such that supply is greater than sediment transport capacity. Under such conditions, sediment transport rates would be proportional to flow, that is, abundant transportable sediment is available and the primary limit on sediment transport is flow magnitude and duration. The channel bed in transport-limited channels is expected to be relatively fine, typically composed of finer gravel and sand with little armoring of the bed surface. Transport-limited channels may be found where there are abundant sediment inputs (e.g. recent concentrated inputs from landslides) or where channel slope declines rapidly (e.g. where a relatively steep confined channel reaches a broad valley with lower channel gradient).

E.3.3.3.2 Supply-limited Channels

Supply-limited channels are defined by high sediment transport capacity relative to sediment supply. Sediment transport rates are high when sediment is available for transport, but relative to the transport-limited condition, the relationship between stream flow and sediment transport is erratic. The channel bed is expected to be relatively coarse, with frequent armoring of bed deposits and frequent bedrock exposures. Although conditions are variable, depending on channel and valley morphology and watershed erosion history, many of the smaller, steeper upland streams important for anadromous fish would be expected to be supply-limited. This expectation is conditioned largely on the high degree of confinement, moderately high slopes, and moderate to intense storm runoff typical of such streams (i.e. factors suggestive of high sediment transport capacity).

Climatic variability is also an important temporal factor in that, during periods of low frequency of intense rainstorms (regional-decadal scale), sediment transport capacity could be significantly reduced. This could conceivably shift channel conditions toward transport-limited from supply-limited in systems or reaches where sediment supply and transport capacity are relatively balanced.

E.3.3.4 Spatial and Temporal Relationship Between Sediment Input Mechanisms and Sediment Transport Phenomena

Coarse sediment is approximately equivalent to bedload sediment excluding the sand that is transported in intermittent suspension and which is thought to be detrimental to spawning habitat. Although the precise size range corresponding to coarse sediment varies among observers and objectives, coarse sediment referred to in this Section is considered to be > 2 mm diameter, and includes gravel, cobbles and boulders.

Landslides are generally the major sources of coarse sediment. Shallow rapid landslides (debris slides and debris flows) generally include significant proportions of coarse sediment, depending on the proportion of gravel in the displaced soil and colluvium. Deep-seated landslides (translational/rotational slides) include coarse sediment from soil and colluvium overlying the slide, as well as coarse sediment derived from the underlying bedrock. Consequently, deep-seated slides have the potential to introduce large sediment clasts (boulders). Even earthflows, which have high proportions of fine sediment inputs (DMG 1997), may introduce very coarse rock that cannot be mobilized by the stream, thus inducing a steepening of the stream channel (Kelsey 1980).

Channel erosion in headwater streams, particularly in mélange (Best et al. 1995), bank erosion, and soil creep processes also introduce coarse sediment to streams in proportion to the concentration of gravel in the soil material. In combination with natural landslides, bank erosion and soil creep are a major source of natural or "background" sediment inputs. Fluvial erosion associated with gullies created by blocked culverts where roads cross streams, or where blocked culverts cause fill failures, may also introduce large quantities of sediment (Best et al. 1995), with the quantity of coarse sediment depending on the proportion of gravel in the soil material. Under past management practices, road construction may have introduced large quantities of sediment to streams as a result of uncontrolled sidecast disposal of soil, or a as a result of poor road construction techniques and poor maintenance. Although such practices are now prohibited, northern California rivers may yet be affected by the legacy of former practices.

The timing and frequency of coarse sediment inputs tend to be dominated by mass wasting processes. With the exception of channel erosion, bank erosion and soil creep, the erosion processes noted above typically generate sediment inputs that are relatively concentrated near the point of entry to the channel network. Landslide deposits in channels typically include abundant coarse and fine sediment and LWD. Deposits may fill existing channels and induce erosion along stream banks. The transport and downstream routing of such coarse sediment budgets have been investigated both in model and field studies of upland rivers (Benda and Dunne 1997a, 1997b; Lisle et al. 1997 and Lisle et al. in press (re: Floodgate slide)). While it is generally agreed that the local effect is greatest at the point of entry, consistent theoretical statements regarding the magnitude and timing of effects downstream and the governing processes are elusive. Benda and Dunne (1997a) hold that concentrated coarse sediment inputs to a channel network are routed downstream in a kinematic wave that persists downstream. Kelsey (1980) observed this phenomenon in the Van Duzen River. In contrast, Lisle et al. (in press) believe that diffusive processes control the routing of sediment, and that such pulses of input gradually disperse downstream. In either case, the greatest short-term effects with respect to coarse sediment are localized, with only gradual (over a period of years to decades) translocation of effects (typically increased depth of gravel deposits and changes in size distribution of bed material).

Landslide inputs of coarse sediment also tend to be concentrated in time in response to periods of unusual precipitation and streamflow. Conditions that are likely to trigger shallow landslides occur relatively infrequently in northern California (Cafferatta and Spittler 1998). The activity of deep-seated landslides tends to be related to longer-term periods of precipitation (Iverson 2000), or to periods of high streamflow that erode toe deposits and destabilize deep landslide blocks.

Debris flows and debris torrents may have broader impact on streams because high concentrations of sediment and woody debris may be carried several thousand feet or more from the initiation site to the distal end of the deposit. Large portions of the affected channel may be scoured to bedrock, while reaches affected by deposition may aggrade substantially. Moreover, in terrain prone to debris flow (e.g. debris slide amphitheater/slope (DMG 1997)), many potential initiation sites may be present in colluvial hollows, creating potential for more frequent debris flow impacts to downstream channels (Benda and Dunne 1997a). Field evidence indicates that episodes of major debris avalanching in headwater channels in northern California probably occurred at intervals of 300 to 2,000 years, and that smaller but significant episodes probably occur more frequently (Kelsey 1982).

E.3.4 Potential Effects on Covered Species

This Section reviews known potential effects of sediment on the Covered Species and the characteristics of their aquatic habitat. The summary of sediment transport and sedimentation processes provided above is applied to give perspective on the relationship between sediment sources and sedimentation effects on habitat. The discussion distinguishes between the effects and sources of “fine” and “coarse” sediment, and is oriented toward conditions found in northern California streams and watersheds.

Although this Section focuses on sediment effects on aquatic habitat, it must be recognized that sediment is not a singular environmental factor affecting habitat conditions. Stream temperature and habitat morphology, particularly in relation to the influence of LWD, are two other major controls on habitat conditions, and both of these have been or potentially may be affected by watershed management. Areas with generous riparian buffers provide a means to recruit LWD and reduce sediment inputs; when mass wasting events occur in such areas, both LWD and coarse sediment will be recruited to channels along with fine sediment. Coarse sediment (in modest amounts) and LWD can both contribute positively to aquatic habitat conditions in the long term (and often in the short term, particularly LWD). In contrast, chronic erosion from roads (road tread surface erosion, small scale mass wasting of road cut slopes, fluvial erosion of ditches and gullies formed by road drainage) contributes fine sediment to streams. Hence, to the extent that fine sediment negatively affects aquatic habitat, erosion from roads is expected to be relatively more likely to degrade aquatic habitat conditions than modest degrees of mass wasting inputs from riparian buffer zones.

E.3.4.1 Coarse Sediment

In the most extreme case, landslide deposits may bury a channel reach to depths sufficient to entomb any organisms present such as larval tailed frogs, southern torrent salamanders and salmonid eggs in redds in the stream bed. More common and widespread effects resulting from increases in bedload sediment supply may also result in channel aggradation and associated decreases in mean channel depth, decreases in pool depth and more mobile, less stable channels, reducing the quantity of rearing habitat for juvenile salmonids and potentially reducing emergence from redds (Bisson et al. 1992, Sullivan et al. 1987). If water temperatures are not increased, aggradation of the channel due to coarse sediment inputs potentially would have less of an impact on the amphibian Covered Species, because they select for riffle habitat and are generally not found in pools (Diller and Wallace 1996, 1999; Welsh and Lind 1996). Coarse sediment inputs of competent material with a small fraction of fines may actually be beneficial to southern torrent salamanders. Material of this type contains an extensive interstitial network through which the salamanders can move.

Effects of excess coarse sediment on pool habitat are believed to be potentially significant for the salmonid Covered Species. Pool abundance and depth has been positively correlated with salmon and trout abundance and density (Bisson et al. 1982; Murphy et al. 1986). Juvenile coho salmon as observed in Green Diamond’s summer population estimates are found almost exclusively within pool habitats in Plan Area streams (Appendix C7). Pool habitats provide summer rearing habitat, and may act as cool water temperature refugia in the summer (Steele and Stacy 1994). The input of coarse bed materials can result in both increased and decreased rearing capacity for juvenile salmonids (Hicks et al. 1991). Coarse sediment inputs have the potential to negatively impact the fish Covered Species through infilling of pool habitat and the localized burial of redds. Such habitat modification could constitute a take of salmonids if it interfered with the ability of those present to shelter or if it destroyed their eggs.

The relatively slow rate of transport of bedload sediment results in relatively persistent effects, depending on local transport rates and the magnitude of the effect. The slow movement of bedload sediment and the tendency for bedload inputs to be concentrated in space in association with landslides suggests that coarse sediment effects may frequently be localized, affecting stream reaches rather than entire watersheds. With the passage of time, assuming inputs of coarse sediment are reduced, negative effects of coarse sediment on salmonid habitat can be expected to dissipate (Sullivan et al. 1987).

E.3.4.2 Fine Sediment

There are two size fractions of fine sediment to consider, each with different effects on habitat. First, there is the intermittent suspended load comprised primarily of fine gravel and sand. This is distinguished from the suspended load/wash load fraction comprised of fine sand, silt and clay particles.

The erosion sources that supply fine sediment to streams include those identified for coarse sediment, however, they also include significant quantities generated by rainsplash, sheetwash, rill, and gully erosion processes occurring primarily on roads and skid trails.

The timing and frequency of fine sediment inputs are potentially distinct from timing and frequency of coarse sediment inputs. Both coarse and fine sediment inputs resulting from landslides tend to be concentrated in time and space. More dispersed and chronic inputs of fine sediment are likely, however, owing to widely dispersed sources and the high frequency of rainfall-runoff events capable of mobilizing fine sediment from sources areas, particularly roads. Most rainstorms are likely to provide sufficient energy to erode and deliver available sediment from road surfaces to streams. Hence, even in relatively dry years when mass wasting processes are insignificant, substantial road surface erosion would occur. Given the propensity for landslide events to be triggered during relatively intense rainstorms, mass wasting episodes tend to be concentrated in a few years over periods of decades at the watershed scale. During the intervening years of relatively low mass wasting, erosion of fine sediment from roads would likely be persistent, potentially magnifying its impact on aquatic habitat.

As described above, wash load and suspended load travel at velocities similar to average stream velocities. Consequently, suspended sediment effects are transient, but may be persistent if the erosion source is persistent. The intermittent suspended load travels at substantially lower velocities, but is nevertheless significantly faster than coarse bedload. Consequently, fine sediment entering the stream system is rapidly dispersed far downstream, and sand and fine gravel deposits on the bed surface can be routed through channel reaches relatively quickly (Lisle and Hilton 1999).

The effects of increased fine sediment input on the Covered Species vary with sediment particle size. Increased inputs of fine sediments are associated with increased embeddedness of spawning substrates and high turbidity levels (Chapman 1988). Increases in fine sediments deposition into stream gravels can lead to a reduction in spawning success, reduced food production, and loss of benthic cover for over-wintering juveniles (Hicks et. al. 1991; Wood and Armitage 1997). The larvae and adults of the southern torrent salamander and larval tailed frogs utilize the interstices within gravel and cobble substrate, and are not typically found in sandy or silty streams (Bury and Corn 1988; Diller and Wallace 1996, 1999). Salmon and trout spawn in gravel and cobble substrates, and sedimentation or burial of these substrates would likely result in reduced reproductive success for these species (Chapman 1988). Subsurface flow through redds is essential in providing dissolved oxygen to embryos and carrying away metabolic wastes. Sedimentation can reduce the survival to emergence of the covered embryos by reducing subsurface flow, and by creating a sediment 'cap' which prevents hatched fry from emerging (Reiser and White 1988). Accordingly, increased embeddedness caused by increased input from Covered Activities could result in take of salmonids by destroying eggs or fry. Laboratory studies have demonstrated that increases in fine sediment in redds reduces survival to emergence either by entombment or by reducing the supply of oxygenated water to the redd, but field experiments have found more variable effects depending on the experiment, region and other environmental factors (Everest et al. 1987).

As noted above, there are several potential habitat effects associated with the coarser fraction of fine sediment (i.e. sand and fine gravel, intermittent suspended load). These include infiltration of fine sediment into coarse sediment that degrades the quality of spawning habitat. Infiltration of sand and fine gravel in coarser gravel and cobble streambeds has been investigated in both laboratory (Carling 1984) and field (Lisle 1989) studies and show that infiltration rate is proportional to sediment transport rates in the stream. Hence, reductions of fine sediment inputs are expected to result in improved spawning conditions for all Covered Species. The coarser fraction of fine sediment has also been found to collect in pools in some stream systems, reducing the quantity and quality of summer rearing habitat and winter refugia in pools (Lisle and Hilton 1999). The extent of pool filling by fine sediment appears to be related to watershed sediment supply.

Additional effects of excessive sediment inputs of either size class on aquatic habitat include aggradation of stream channels and loss of bank stability, resulting in a wide, shallow channel with low canopy cover, higher water temperatures, and intermittent surface flows in low flow conditions (Swanston 1991). These secondary effects are typically seen in the depositional reaches of streams, making them likely to impact the salmonids but not the amphibian Covered Species.

The finer fraction of fine sediment, primarily silt and clay transported in suspension in water column (suspended load and wash load) is highly correlated with turbidity. High levels of suspended sediment have been found, primarily in laboratory experiments, to have a range of deleterious effects on salmonids. An increase in fine sediments can also lead to chronic levels of turbidity, which may damage the gills of salmonids, reduce their growth rate, impair the ability of fish to locate food, and negatively impact the macroinvertebrate production (Bozek and Young 1994; Sigler et. al. 1984; Newcombe and MacDonald 1991). Negative effects of suspended sediment on juvenile salmonids depend on sediment concentration and duration of exposure, and the interaction of these factors is not well understood (Newcombe and MacDonald 1991). In addition, the availability of localized refugia from high suspended sediment concentrations, such as side channels and backwater pools, may also affect both concentration and duration of exposure. Gregory (1993) indicated that suspended sediment may have some beneficial effects as well, such as providing cover from predators. Thus, fine sediment inputs from the Covered Activities could take salmonids by impairing their ability to breathe, grow and eat.

It is not known if there are any direct effects of increased suspended sediment or turbidity on the amphibian Covered Species. Green Diamond speculates that it has the potential to impact the aquatic dependent larval stages of these amphibians in the same manner as was noted above for the salmonids. In addition, suspended sediments could influence the growth of diatoms on the stream's substrate, which is the sole food for larval tailed frogs. Southern torrent salamanders are less likely to be impacted by suspended sediments, because they occur in seeps, springs and the uppermost reaches of streams that are generally not influenced by the downstream transport of fine sediments. However, Green Diamond believes that it is more likely that increases in suspended sediment (especially the larger particle sizes) would impact the amphibians indirectly by reducing interstices in the substrate and causing substrate embeddedness.

E.3.4.3 Potential Severity of Effects of Suspended Sediment on Salmonids

Newcombe and Jensen (1996) developed a model based on results of previous experiments for the effect of suspended sediment on salmonids in terms of concentration of suspended sediment and duration of exposure. They developed a concentration-exposure function that predicts the severity of the effect on adult and juvenile salmonids. To gain perspective on the effects of suspended sediment on salmonids in the study area, a series of paired suspended sediment and discharge observations representing hydrographs of peak runoff at eight USGS stream gauging stations in northern California with drainage areas ranging from about 1.5 to 30 square miles was evaluated with respect to predicted severity of effects according to Newcombe and Jensen's model.

Subsequent work by Newcombe and Jensen (1996) developed a model to predict quantitatively the effects of elevated suspended sediment on salmonids. This model was employed to evaluate available regional data and assess the potential magnitude of effects of suspended sediment on salmonids summarized below. The objective of this analysis was to develop quantitative perspective on the effects of suspended sediment on salmonids, particularly coho salmon, according to Newcombe and Jensen's (1996) model predicting severity of effects.

E.3.4.4 Methods-Hydrograph Analysis

Nine gauged sites were sampled from existing regional data that was readily available from internet data libraries. A series of paired suspended sediment and discharge observations representing hydrographs of peak runoff at nine USGS stream gauging stations in northern California with drainage areas ranging from about 1.4 to 30 square miles were evaluated (Table E-3). The data was collected in the 1970s, 1980s and early 1990s. Sites with at least 100 paired observations were chosen in order that a relatively large set of potential hydrographs were available for evaluation.

In order to compare discharges among different gage sites, discharge data were divided by the site's drainage area, thus producing unit discharge data (cfs/mi^2). Unit discharge data also provide perspective regarding approximate flow recurrence interval for the flows evaluated. Regional values of the 2 yr recurrence interval event range from about 40 to 80 cfs/mi^2 for streams with the range of drainage areas for these sites (Table E-3).

Table E-3. Summary of USGS suspended sediment gauging stations.*

Station #	Station Name	Area (mi^2)	# Obs	First	Last
11482110	Lacks C Nr Orick Ca	16.9	224	11/22/74	03/20/91
11482125	Panther C Nr Orick Ca	6.1	108	01/12/79	01/04/91
11482130	Coyote C Nr Orick Ca	7.8	100	12/11/78	05/15/89
11482225	Harry Wier C Nr Orick Ca	3.0	169	11/07/73	02/20/80
11482260	Miller C A Mouth Nr Orick Ca	1.4	134	11/07/73	01/22/81
11482450	Lost Man C Nr Orick	4.0	124	10/23/73	02/20/80
11482468	Little Lost Man C A Site No 2 Nr Orick Ca	3.5	192	03/29/74	05/11/89
11530020	Supply C A Hoopa Ca	15.9	123	11/03/81	01/01/84
11532620	Mill C Nr Crescent City	28.6	107	01/16/74	12/24/80

Note

* Of these nine stations, only the Little Lost Man Creek had little historic commercial forestry in its watershed.

After each sites' data were reviewed, data were selected for more intensive sampling periods during peak flows where at least 3 samples were taken in no more than four days; most of the selected data had several samples during a period of up to 4 days. For selected periods of flow, observations of suspended sediment concentration and unit discharge were plotted against time to generate sedigraphs. An example of one of these more intensively sampled events is shown in Figure E-1.

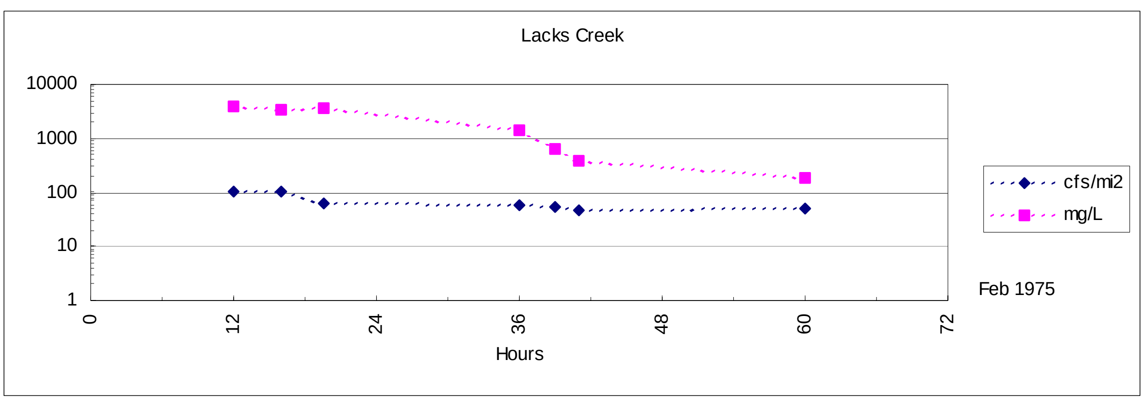

Figure E-1. Example of a more intensively sampled site in Lacks Creek during February 1975.

In some cases, these sedigraphs were then decomposed into shorter time periods based on interpretation of the hydrograph. The intent was to identify discrete storm hydrographs where suspended sediment concentrations appeared to reach sustained high levels, representing a 'worst case' scenario. Within these shorter time periods, the average suspended sediment concentration and unit discharge was calculated (Table E-4). The peak suspended sediment discharge and peak unit discharge also was noted. The time duration was determined by subtraction between the first and last observation.

E.3.4.5 Methods-Severity of Effects

Newcombe and Jensen (1996) reviewed 80 studies that documented fish responses to suspended sediment. Their analysis resulted in six empirical equations that quantify the biological response of fishes to duration of exposure and suspended sediment concentration. The authors quantified fish responses by creating the severity of ill effect (Table E-3).

The equation used to determine the SEV is:

$$Z=a +b(\log_e x) + c(\log_e y)$$

where z is the severity of ill effect, x is duration of exposure (in hours), y is concentration of suspended sediment (mg SS/L), a is the intercept and b and c are slope coefficients (Newcombe and Jensen 1996).

The natural log of the duration of exposure and the suspended sediment concentration were calculated and rounded to the nearest whole number. From there, the tables in Newcombe and Jensen (1996) were used, specifically Figure 1 (juveniles and adult salmonids), Figure 2 (adults salmonids only), and figure 3 (juvenile salmonids only). . Each of these figures presents SEV values for a given suspended sediment concentration and duration. Additionally, data from the appendix also were used to examine SEV values specifically for underyearling coho salmon (see Figure E-2). Similar curves were constructed for smolt and juvenile coho life stages, but the distribution of data was quite uneven and was difficult to interpret directly (unlike the data in Figure E-2).

E.3.4.6 Results

This analysis indicated that the worst case effects were “paralethal” (SEV=9). These conditions existed in Lacks Creek in February 1975 as well as in February 1979, and in Supply Creek near Hoopa in April 1982. All other scenarios indicate sublethal SEV values between 5 and 8.

Table E-3. Potential Severity of Effects (SEV) of suspended sediment on salmonids at nine gauged stations in Northern California.*

Gage Station	Hydrograph Dates	Mean Discharge (cfs/mi^2)	Peak Discharge (cfs/mi^2)	Mean SSC (mg/L)	Max. SSC (mg/L)	Hydrograph Duration (hours)	SEV Case 1	SEV Case 2	SEV Case 3	SEV Case 4
Lacks Creek Near Orick	2/12/1975 2/13/1975	80	102	3115	3900	24	9	9	9	8
	2/13/1975 2/14/1975	65	53	407	658	21	7	8	7	6-7
	2/10/1979 2/11/1979	8	11	127	159	24	7	7	6	4
	2/12/1979 2/13/1979	26	51	734	1140	34	9	9	9	6-7
Harry Wier Creek Near Orick	11/7/1973 11/8/1973	45	94	754	2070	11	7	8	7	7
	11/8/1973 11/9/1973	35	49	348	646	27	7	8	7	5-6
	3/1/1974 3/2/1974	17	21	148	377	19	7	7	6	5
Miller Creek Near Orick	11/7/1973 11/8/1973	46	76	1644	2730	13	8	8	8	8
	11/8/1973 11/9/1973	26	32	404	557	21	7	8	7	6-7
	2/12/1975 2/13/1975	29	32	1340	1530	11	7	8	7	7
	2/13/1975 2/14/1975	26	32	259	495	24	7	8	7	6-7
Lost Man Creek Near Orick	11/7/1973 11/8/1973	43	55	1089	1790	12	7	8	7	7
	11/8/1973 11/9/1973	40	53	224	388	27	7	7	6	5-6
Supply Creek Near Hoopa	4/11/1982 4/15/1982	30	49	337	822	98	9	9	9	5-6
Mill Creek Near Crescent City	3/17/1975 3/18/1975	65	84	286	406	16	7	8	7	6-7
	3/18/1975 3/19/1975	133	153	1119	1450	26	8	8	8	7
Panther Creek Near Orick	3/14/1980 3/15/1980	37	46	328	407	27	7	8	7	5-6
	1/15/1988 1/16/1988	25	29	200	258	23	7	7	6	6
Coyote Creek Near Orick	3/13/1980 3/14/1980	56	61	1783	1880	26	8	8	8	7
	3/15/1980 3/17/1980	24	29	228	296	50	7	7	7	5-6
Little Lost Man Creek Near Orick	3/18/1975 3/18/1975	185	192	2455	2830	1	7	8	6	6

Notes

* SEV values between 4 and 8 are considered "sublethal" and values equal to or greater than 9 are considered as "paralethal to lethal" (see Table E-3). SEV values for Cases 1 through 4 are shown; see below for definitions of cases.

SEV 1 - Adult and Juvenile Salmonids (Figure 1, Newcombe and Jensen 1996)

SEV 2 - Adult Salmonids (Figure 2, Newcombe and Jensen 1996)

SEV 3 - Juvenile Salmonids (Figure 3, Newcombe and Jensen 1996)

SEV 4 - Underyearling Coho (Appendix, Newcombe and Jensen 1996)

Table E-4. Scale of severity of ill effects associated with excess suspended sediment.

SEV	Description of effect
0	No behavioral effects
1	Alarm reaction
2	Abandonment of cover
3	Avoidance response
4	Short term reduction in feeding rates; short term reduction in feeding success
5	Minor physiological stress; increase in rate of coughing, increased respiration rate
6	Moderate physiological stress
7	Moderate habitat degradation; impaired homing
8	Indications of major physiological stress; long term reduction in feeding rate, long term reduction in feeding success; poor condition
9	Reduced growth rate; delayed hatching, reduced fish density
10	0-20% mortality; increased predation; moderate to severe habitat degradation
11	>20-40% mortality
12	>40-60% mortality
13	>60-80% mortality
14	>80-100% mortality

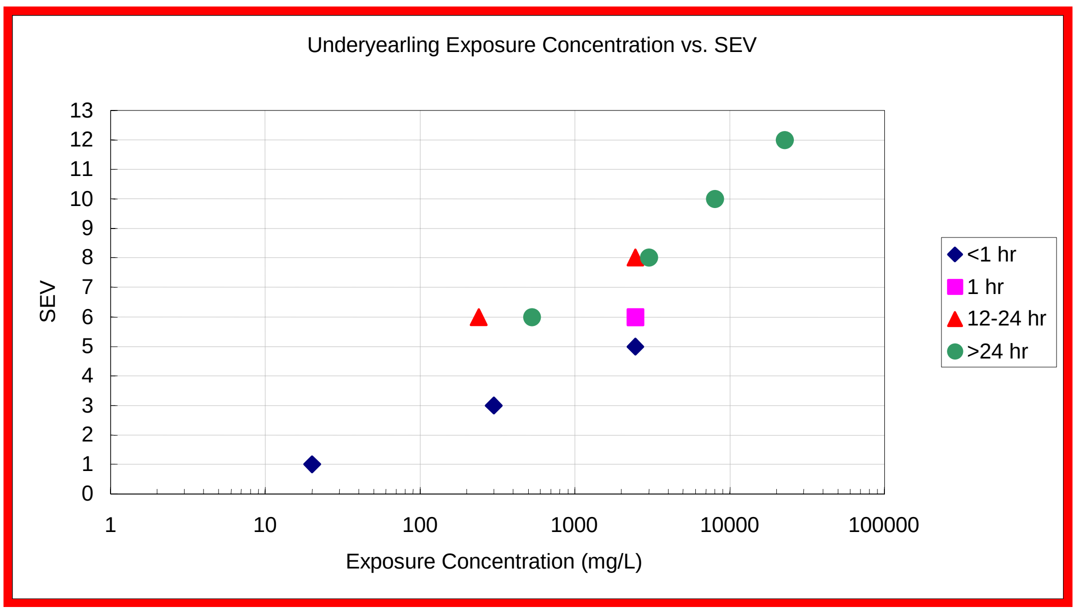

Figure E-2. Data from Newcombe and Jensen's (1996) appendix for "underyearling" coho salmon (SEV Case 4 in Table E-2).

E.3.4.7 Discussion

The suspended sediment data from all but one of the sites evaluated presumably reflect the effects of substantial pre-Forest Practices Act logging practices in watersheds that were managed largely for timber production. Consequently, these data represent relatively poor watershed conditions, probably substantially worse with respect to management-induced erosion than would be found under current conditions.

The results of this analysis show that these streams experience periods of elevated sediment concentrations that are predicted to induce physiological stress on salmonids. This suggests that such stressing conditions are likely to be present to some degree in many northern California streams. Given regional erosion rates (see section E.3.2 above), it may well be that these stressing conditions occur regardless of management influences.

Presumably, forest management would tend to increase the frequency and/or magnitude of these stressing events. The long-term effects of these stressing events on salmonids are not well known. It has been suggested that extended periods of higher turbidity (generally correlated with higher suspended sediment concentration in the region), could interfere with feeding success of juvenile salmonids, reducing the size of smolts, which in turn would presumably reduce survival rates in the oceanic life-stage. In this analysis, data for juvenile coho salmon (see Figure E-2) indicate a somewhat lower SEV score (SEV Case 4) than for salmonids in general. This suggests the possibility that coho salmon may be somewhat better adapted to cope with suspended sediment. Given the typical geologic conditions in the coastal watersheds where these fish evolved, this possibility appears plausible. No similar studies have been done for the amphibian Covered Species to quantify the impact of suspended sediments on any life stage.

Sediment inputs, both coarse and fine, are absolutely essential to maintain a healthy lotic system. However, excess sediment inputs can have diverse and highly negative impacts. As described in the discussions above, the potential impacts from increased sediment inputs vary depending on the primary particle size involved (i.e. coarse versus fine). The impacts are generally cumulative in nature, especially for the finer particle sizes that can stay suspended in the water column and potentially impact regions at great distances downstream of the sediment source. The life history stage of the Covered Species that are potentially impacted by various types of sediment inputs is also variable, but there is the potential for all life history stages to be negatively impacted in a manner resulting in take. Increased sediment inputs can produce a myriad of negative impacts on habitat, such as increased pool filling, embeddedness, increased temperature and turbidity can potentially result in direct mortality, and decreased survival rates of various life history stages of the Covered Species, particularly in early life stages. Such impacts, and more importantly, changes in population demographic parameters, may result in local population declines. Such declines could negatively affect the regional populations of the Covered Species.

E.3.5 Altered Thermal Regimes

E.3.5.1 Altered Riparian Microclimate

The riparian microclimate has potentially important indirect effects on the salmonid Covered Species and aquatic forms of the amphibian Covered Species through alteration of water temperature, which will be discussed in the following Section. However, the riparian microclimate also has potentially important direct effects on the adult forms of the amphibians. Reduction of riparian overstory canopy through timber harvesting could result in increased levels of incident solar radiation reaching the stream and riparian zone during the day and reduced thermal cover at night (Welch et al. 1998). It could also increase exposure to wind in the riparian areas due to an edge effect from an adjacent harvest unit with the overall net effect of increasing daily fluctuations in air temperature and relative humidity. Studies done in areas outside the coastal influence of the 11 HPAs indicate that microclimatic edge effects can be detected as much as 240 meters (787 feet) from the edge of a clearcut (Chen 1991). However, the greatest attenuation of edge effects on microclimatic changes occurs within the first 30 meters (98 feet) of the buffer (Ledwith 1996). Although the impact of altered riparian vegetation on the microclimate is ameliorated by the cool coastal climate in the region, reduction of riparian cover due to timber harvesting has the potential to cause greater daily and seasonal fluctuations in the microclimate of the riparian areas.

In addition, increased coarse sediment inputs from management activities, particularly when it occurs in the form of debris torrents, can result in widening of the channel and loss of streamside vegetation (Swanston 1991). Just as in overstory canopy loss, this has the potential to alter the riparian microclimate by increasing daily fluctuations in air temperature and relative humidity. It is unlikely that increases in air temperature with corresponding decreases in relative humidity during the day would directly impact the amphibians, because the adults are not surface active during the day. However, the corresponding drying effect of increased air temperature and decreased relative humidity could result in the loss of some daytime refugia habitat and nighttime foraging sites. It is also possible that the reduction of thermal cover at night may impact the ability of adults to forage at night.

E.3.5.2 Altered Water Temperature

Loss of riparian overstory canopy through timber harvesting and increased coarse sediment inputs from management activities could result in alteration of the riparian microclimate as described above. However, changes in the riparian microclimate will also result in corresponding changes in the daily water temperature regime. In addition, both reduction of overstory canopy and increased coarse sediment inputs can result in altered water temperature through direct mechanisms. Removal of the riparian canopy will result in elevated summer water temperatures, often in direct proportion to the increase in incident solar radiation that reaches the water surface (Chamberlain et al. 1991). For a given exposure from solar radiation, water temperature increases directly proportional to the surface area of the stream and inversely proportional to stream discharge (Sullivan et al. 1990). Exposed channels will also radiate heat more rapidly at night. In addition, increased sediment inputs that results in aggradation will result in a wider and shallower channel that gains and losses heat more rapidly. Therefore, reduction of riparian vegetation and aggradation of a channel act synergistically to cause greater daily and seasonal fluctuations in water temperatures.

While the increases in summer water temperatures may be large after removal of riparian vegetation, the changes in winter water temperatures are usually less dramatic. However, slight changes in temperature may have a large impact on salmonids when water temperatures tend to be low. Studies on a coastal watershed in British Columbia revealed that the number, size and migration timing of coho smolts were most affected by small increases in late-winter and early-spring water temperatures (Hartman et al. 1987). Generally, the removal of riparian vegetation resulted in increases of winter water temperatures in low elevation coastal watersheds due to increases of solar energy (Beschta et al. 1987). Conversely, in northern latitudes and at higher elevations decreases in winter water temperatures may occur due to the loss of insulation from riparian vegetation, leading to an increase in radiative cooling from the watershed.

Changes in water temperatures from the removal of riparian vegetation may benefit or negatively impact salmonid populations. Among the potential benefits is an increase in primary and secondary production that would increase the amount of available food. Studies have reported that after logging, increases in filamentous algae promoted the abundance of invertebrate grazers such as baetid mayflies, grazing caddisflies and midges that were more likely to contribute to the drift and be available as food for salmonids (Hawkins et al. 1982). Increased water temperatures during winter months are usually less dramatic than summer increases; however these slight increases may have a great effect on salmonids. Studies conducted on Carnation Creek in British Columbia revealed that slight increases in winter water temperatures resulted in accelerated development of coho embryos, thus an earlier emergence of juveniles (Hartman et al. 1987; Holtby 1988). The earlier emergence resulted in a longer growing season for the juvenile coho salmon, but also increased their risk to downstream displacement during late-winter storms. The increased growth of juvenile coho resulted in higher over-wintering survival rates of 1+ fish. However, in Carnation Creek the out-migration of coho smolts was highly correlated with spring temperatures, thus the slightly elevated temperatures resulted in an earlier out-migration (Hartman et al. 1987). These early migrants probably reached the estuarine environment when conditions were not favorable for smolt survival. Additional studies predicted that the marine survival of 1+ coho smolts declined from 14.3% to 10.7% and the marine survival of 2+ coho smolts declined from 15.6% to 10.7% (Bilton et al. 1982). Apparently earlier migration into unfavorable marine conditions negated any survival advantage of increased smolt growth due to increases in water temperature (Hartman et al. 1987).

Increased water temperatures can also have negative impacts on the salmonids (Beschta et al. 1987) as well as the amphibians. Potential impacts to salmonids from increased stream temperatures include (Hallock et al. 1970; Hughes and Davis 1986; Reeves et al. 1987; Spence et. al. 1996):

- reduction in growth efficiency,
- increased disease susceptibility,
- changes in age of smotification.
- loss of rearing habitat, and
- shifts in the competitive advantage of salmonids over non-salmonid species.

There is a potential secondary negative impact of increased water temperatures that is related to levels of dissolved oxygen in the water. During summer months, low flows and increased water temperatures accelerate respiration and reduce the solubility of oxygen. The reduction of available oxygen may reduce growth rates of individual fish and may limit the production capability of an entire watershed.

Although the specific mechanisms are not known, many of the same physiological or ecological factors associated with elevated water temperatures presumably exist for the amphibian species, which have temperature thresholds below those of the fish Covered Species.

Although elevated water temperatures can be a relatively localized phenomenon, this factor generally functions in a cumulative manner throughout a sub-basin or watershed. The impact of elevated water temperature also tends to be cumulative on a temporal scale, such that short-term increases are less likely to be harmful compared to more chronic increases in water temperature. The potential harm or death associated with this factor would primarily influence the juvenile salmonids and larval amphibians during summer and early fall. Take of Covered Species could occur as the result of temperature increases causing the impairment of essential functions and injury or mortality. The potential impacts of such taking include potential reductions in the local or regional populations of the Covered Species and could affect a possible need to list currently unlisted Covered Species under the ESA in the future.

E.3.6 Altered Nutrient Inputs

Unlike lentic systems and the mainstem of many rivers in which runoff from agricultural, suburban, industrial and other areas lead to eutrophication, the portion of lotic systems throughout the Pacific Northwest and Northern California in which salmonids spawn and rear are thought to be naturally oligotrophic due to low levels of nitrogen (Allan 1995; Triska et al. 1983). However, additions of nitrogen in these systems will only result in limited increases in primary productivity, because most of these streams, especially heavily shaded lower order channels, are also limited by light (Triska et al. 1983). While autochthonous inputs (derived from within the aquatic system through photosythesis) are important in higher order channels, much of the energy and nutrients in lower order channels (where many salmonids rear) comes from allochthonous inputs (derived from outside the aquatic system typically through detrital inputs). One of the most important sources of detrital inputs in streams throughout the Northwest comes from red alder, because it is readily available to the aquatic invertebrate community and its leaves are high in nitrogen (Murphy and Meehan 1991; pers. comm. K. Cummins, Humboldt State University). The fact that red alder fixes atmospheric nitrogen also has important implications for increasing the total available nitrogen in these potentially oligotrophic lotic systems. In contrast to red alder leaves that can be 50% decomposed in less than 2 months, Douglas-fir needles may take over 9 months to reach the same level of decay and have far less nitrogen. Woody debris, even twigs and small branches, has limited nutritional value to streams because it decays so slowly and is very low in nitrogen (Murphy and Meehan 1991). Another potentially important source of nutrients to streams comes from annual spawning runs of anadromous salmonids. Reduced ocean-derived nutrients to stream and riparian ecosystems due to declines in salmon returns in many regions have received considerable attention in recent years (AFS: Nutrient Conference 2001). This has lead to numerous studies looking at the potential benefits of artificially increasing the productivity ("jump-starting") of these systems through the addition of salmon carcasses or other sources of nutrients.

Reduction of riparian vegetation due to timber harvest is likely to increase productivity of streams in several ways. Increased incident solar radiation would likely increase periphyton production (unless it is limited by nitrogen), which may increase the abundance of invertebrates and fish due to an enhanced quality of detritus. The mechanism of this increase is tied to the algae, a higher quality food than leaf or needle litter, which increases the abundance of invertebrate collectors, which in turn, can increase the abundance of predators such as juvenile salmonids (Murphy and Meehan 1991). In addition, timber harvest in riparian areas may reduce the number of conifers and increase deciduous vegetation such as red alder. Therefore, with increased input of nutritionally rich leaf detritus compared to conifer needles, productivity of the stream may increase. Of course, the salmonid response would only be realized if the alteration of the riparian vegetation did not also lead to adversely high water temperatures. An increase in stream productivity may also not ultimately result in increased production of salmonids, because it will primarily benefit summer rearing populations when the "bottleneck" (i.e. limiting factor) for many salmonid streams is winter rearing habitat (Murphy and Meehan 1991).

Larval tailed frogs feed exclusively on diatoms that grow on the surface of the stream's substrate (Metter 1964). Growth of the diatoms is influenced by factors such as sunlight, water temperature and nutrients, but there have been no studies to determine if diatomaceous growth is ever limiting for larval tailed frogs. As a result, it is not possible to speculate on how altered nutrients may influence this life history stage of tailed frogs. The adult frogs presumably feed in the riparian zone, but there is little known of their foraging ecology and it would not be possible to speculate on how altered nutrients in the stream might influence the adults. Larval and adult southern torrent salamanders feed primarily on small aquatic invertebrates whose numbers would be influenced by detrital inputs. However, it is not known if food is very limiting for this species such that changes in aquatic invertebrates would influence survival or growth of individual salamanders.

The impacts of altered nutrient inputs would most likely be subtle and difficult to predict. The greatest potential impact would be to juvenile salmonid populations that need to reach some threshold in size before smoltification and out-migration can occur. Decreases in nutrient inputs would not likely result in direct harm, but they may reduce survival during the freshwater rearing period. In addition, ocean survival would likely be decreased if smolts out-migrate at smaller sizes.

E.4 LWD RECRUITMENT AND DISTRIBUTION

Historically, the mainstems of watersheds were utilized to transport logs downstream to processing mills. Thus, extensive clearing of debris jams occurred on most coastal watersheds (Sedell and Froggart 1984). Splash damming was another management technique to transport logs downstream that tended to dislodge established LWD from stream channels. These channel clearing activities directly removed salmonid habitat from watersheds and also reduced the probability of additional LWD retention within the channel.

Inchannel salvage logging and the clearing of LWD from streams in the Pacific Northwest began shortly after the 1964 Flood. Much of this activity was sponsored by the federal government as a measure to protect bridges and to reduce cases of property liability in court (Maser and Sedell 1994). Removal of LWD from stream channels also occurred during the 1970s and 1980s when state and federal agencies spent over six million dollars annually in efforts to remove debris jams and improve fish habitat (Maser and Sedell 1994). Many of the large debris jams were probably barriers to fish migration and required modification. However, these stream clearing programs often went too far and now fisheries managers have spent the past 15 years reintroducing LWD to streams along the Pacific Northwest. Currently, some fisheries biologists consider the placement of LWD restoration structures in streams as an interim, short-term measure until large conifers are reestablished in riparian zones to provide a source of LWD (House et al. 1989).

Decades of timber harvesting in the riparian zone has altered the species composition and age classes of trees along stream channels. The removal of valuable conifer species has led to the predominance of early successional species such as alders and willows. Short-rotation harvesting has decreased the numbers of large trees available as potential LWD. Woody debris from second-growth forests has a shorter residence time in stream channels than debris from uncut watersheds (Grette 1985). Managed riparian zones of predominately red alder may have a greater input rate of wood to the stream channel than conifers in an uncut riparian zone, but the reduced longevity of alder debris results in reduced cover and fewer pools than in uncut watersheds (Grette 1985).

In-channel LWD is recognized as a vital component of salmonid habitat, and to a lesser extent, but still important to the amphibian Covered Species. The physical processes associated with LWD include sediment sorting and storage, retention of organic debris, and modification of water quality (Bisson et al. 1987). The biological functions associated with LWD structures include important rearing habitats, protective cover from predators and elevated stream flow, retention of gravels for salmonid redds, and regulation of organic material for the instream community of aquatic invertebrates (Murphy et al. 1986; Bisson et al. 1987). Decreased supply of LWD can result in (Hicks et. al. 1991 as cited by Spence et al. 1996):

- reduction of cover,
- loss of pool habitats,
- loss of high velocity refugia,
- reduction of gravel storage, and
- loss of hydraulic complexity.

These changes in salmonid habitat quality can lead to increased predator vulnerability, reduction of winter survival, reduction in carrying capacity, lower spawning habitat availability, reduction in food productivity and loss of species diversity.

In headwater streams, LWD is also known to dissipate hydraulic energy, store and sort sediment, and create habitat complexity (O’Connor and Harr 1994). Creating and providing cover for pools, a primary function of LWD for salmonids, may be of limited benefit to the headwater amphibian Covered Species since torrent salamanders and larval tailed frogs prefer riffle habitats (Diller and Wallace 1996, 1999; Welsh and Lind 1996). The primary benefit of LWD to the amphibians is the creation of suitable riffle habitat through the storing and sorting of sediment. In addition, LWD that is perched a short distance above the streambed will often form a dam composed of coarse sediment and small woody debris through which water percolates. In streams that are otherwise too embedded with fine sediments to be used by torrent salamanders, this appears to form the only habitat that still supports the species (Diller, pers. comm.). There is circumstantial evidence that these same sites are utilized for egg laying by tailed frogs, but searching such sites is too destructive to adequately investigate the phenomenon (Diller, pers. comm.).

The decline of recruitment of potential LWD from riparian zones can be expected to reduce LWD recruitment to streams for decades following timber harvest of riparian areas. High in the watershed, the potential impacts would be primarily localized, but in larger streams lower in the watershed, LWD can be transported during higher flow events and the impacts may be cumulative. A decline in pool density, pool depth, instream cover, gravel retention, and sediment sorting are likely to result if LWD recruitment is reduced. These habitat changes may reduce the growth, survival, and total production of salmonids as well as the amphibian species (Steele and Stacy 1994; Murphy et al. 1986). Given that LWD is likely critical to provide habitat and cover for juvenile salmonids in both summer and winter, survival rates of these life history stages may be limited by the amount of LWD in some streams. Such potential impacts that reduce survival rates of key life history stages of the Covered Species may result in local population declines. Such declines could negatively affect the regional populations of the Covered Species.

E.5 CUMULATIVE WATERSHED EFFECTS

In general, cumulative watershed effects (CWEs) can be categorized as incremental changes that induce changes in watershed processes that alone are not overwhelming, yet if combined, the impacts on stream channels and habitat for aquatic species are detrimental. This is largely a theoretical concept without empirical data, because the identification of CWEs is difficult due to both the technical complexities of designing statistically valid field studies, and because few research efforts have been sustained for extended time periods. Recently, efforts have been made to examine the cumulative effects of timber harvesting on salmonid bearing watersheds. For example, the Carnation Creek watershed study in British Columbia was an 18 year project to examine CWEs due to timber harvesting (Hartman et al. 1987). Poulin (1984) created synoptic study designs that could examine CWEs by simultaneously studying numerous watersheds at various stages of timber management. Technological advances such as time series analysis of aerial photography, vegetative dating techniques, sediment analysis, and computer modeling systems also provide information about CWEs (Chamberlain et al. 1991). Likewise, extensive literature searches and reviews of historical media and agency files also have assisted in defining past management treatments and resultant effects on Pacific Northwest watersheds (Sedell and Froggart 1984). Historic documents also were used by Sedell et al. (1991) to investigate the transportation and storage of logs in watersheds and the impacts to channel formations and salmonid habitat.

Records of natural changes also are essential to assessing CWEs. Natural change usually occurs within limits that are "normal" for a particular watershed and the biological communities are usually adapted to those changes on either an individual or population level. However, naturally occurring events can inflict catastrophic change on watersheds. Wild fire, drought, floods, volcanic eruptions and earthquakes can all drastically alter physical and biological watershed processes. These events may force salmonid populations to utilize other habitats and undergo reductions in population numbers until the aquatic habitat recovers. The key to the recovery of biological communities is that catastrophic events occur sporadically and the events may also only impact a certain portion of a watershed. Management practices may or may not allow aquatic ecosystems the time to recover before additional impacts are imposed.

E.6 LITERATURE CITED

Allan, J. D. 1995. Stream Ecology- Structure and function of running waters. 1st ed. Chapman & Hall. xii,388 pp.

American Fisheries Society, 2001. AFS: Nutrient Conference 2001. Eugene, Oregon.

Anderson, H.W. 1971. Relative contributions of sediment from source areas and transport processes, Pages 55-63 in Proceedings of forest land uses and stream environment symposium. J.T. Krygier and J.D. Hall, editors. Con. Ed. Publ., Oregon State University, Corvalis, Oregon.

Benda, L. and T. Dunne. 1997a. Stochastic forcing of sediment supply to channel networks from landsliding and debris flow. Water Resources Research. 33(12): 2849-2863.

Benda, L. and T. Dunne. 1997b. Stochastic forcing of sediment routing and storage in channel networks. Water Resources Research, 33(12): 2865-2880.

Berris, S. N. and R.D. Harr. 1987., Comparative snow accumulation and melt during rainfall in forested and clear-cut plots in the western Cascades of Oregon., Water Resources Research 23: 135-142.

Beschta, R.L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Water Resources Research 14: 1011-1016.

Beschta, R.L., R, E. Bilby, G.W. Brown, L.B. Holtby, and T.D. Hofstra, 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. Pages 191-232 In Streamside management: forestry and fishery interactions. E.O. Salo and T.W. Cundy, editors, Contribution No. 57, College of Forest Resources, University of Washington, Seattle, Washington.

Beschta, R.L., M.R. Pyles, A.E. Skaugset and C.G. Surfleet. 2000, Peakflow responses to forest practices in the western cascades of Oregon.: Journal of Hydrology, v. 233, p. 102-120.

Best, D.W., H. Kelsey, D.K. Hagans and M. Alpert. 1995, Role of fluvial hillslope erosion and road construction in sediment budget of Garrett Creek, Humboldt County, CA: Redwood National Park, CA, in Nolan, K.M., Kelsey, H.M., and Marron, D.C., eds., Geomorphic Processes and Aquatic Habitat in the Redwood Creek Basin, Northwestern California, U.S. Geological Survey Professional Paper 1454, p. M1-M9.

Bilton, H.T., D.F. Alderdice, and T. Schnute. 1982. Influence of time and size at release on juvenile coho salmon (*Oncorhynchus kisutch*) on returns at maturity. Can J. Fish. Aq. Sci. 39:426-447.

Bisson, P.A., J.L. Nielsen, R.A. Palmason, and L.E. Grove. 1982. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low streamflow. Pages 62-73 in Armantrout (1982).

Bisson, P.A., R.E. Bibly, M.D. Bryant, C.A. Dolloff, C.S. Grette, R.A. House, M.L. Murphy, K.V. Koski and J.R. Sedell. 1987. Large woody debris In forested strearms in the Pacific Northwest: past, present and future. Pages 143-190 in Streamside management: forestry and fishery interactlons. E.O. Salo and T.W.Cundy,editors. Contribution No.57. College of Forest Resources, University of Washington, Seattle, Washington.

Bisson, P.A., T.P. Quinn, S.V. Gregory, and G.H. Reeves. 1992. Best management practices, cumulative effects, and long-term trends in fish abundance in Pacific Northwest river systems. In R.J. Naiman, ed., Watershed Management: Balancing Sustainability and Environmental Change. New York: Springer-Verlag.

Bosch, J.M., and J.D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J. Hydrol., 55: 3-23.

Bozek, M.A., and M.K. Young. 1994. Fish mortality resulting from delayed effects of fire in the Greater Yellowstone Ecosystem. Great Basin Nat. 54: 91-95.

Brown, W.M. and J.R. Ritter. 1971. Sediment Transport and Turbidity in the Eel River Basin, California. Geological Survey Water-Supply Paper.1986. United States Department of the Interior.

Buffington, J.M. and D.R. Montgomery. 1999. Effects of hydraulic roughness on surface texture of gravel-bed rivers. Water Resources Research 35(11): 3507-3521.

Burroughs, E.R., Jr. and B.R. Thomas, 1977. Declining root strength in Douglas-Fir after felling as a factor in slope stability, U.S. Dept. of Agriculture, Forest Service Research Paper INT-190, 27p.

Bury, R.B., and P.S. Corn. 1988. Douglas-fir forests in the Oregon and Washington Cascades: relation of the herpetofauna to stand age and moisture. Pages 11-22 in R. C. Szaro, K. E. Severson, and D. R. Patton, technical coordinators. Management of amphibians, reptiles, and small mammals in North America. U.S. Forest Service General Technical Report RM-GTR-166

Cafferata, P.H., and T.E. Spittler. 1998. Logging Impacts of the 1970's vs. the 1990's in the Casper Creek Watershed: USDA Forest Service Gen. Tech. Rept., p. 103-115.

Carling, P.A. 1984. Deposition of fine and coarse sand in an open-work gravel bed. Can. J. Fish. Aquat. Sci. 41: 263-270.

Cashman, S.M., H.M. Kelsey, and D.R. Harden. 1995. Geology of the Redwood Creek Basin, Humboldt County, California: Geomorphic Processes and Aquatic Habitat in the Redwood Creek Basin, Northwestern California. U.S. Geological Survey Professional Paper 1454-B, p. B1-B13.

Cederholm, C.J., L.M. Reid and E.O. Salo. 1980. Cumulative effects of logging road sediment on salmonid populations in the Clearwater River, Jefferson County, Washington, College of Fisheries, University of Washington, Seattle, Washington. Contribution No. 543: 35 pp.

Chamberlain, T.W., R.D. Harr, and F.H. Everest. 1991. Timber harvesting, silviculture and watershed processes. Pages 181-205 *in* Influences of forest and rangeland management on salmonid fishes and their habitats. W.R. Meehan, editor. Am. Fish. Soc. Spec. Pub. 19, Bethesda, Maryland.

Chapman, D.W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Trans. Am. Fish. Soc. 117: 1-21.

Chatwin, S.C., D.E. Howes, J.W. Schwab and D.N. Swanston. 1994. A guide for the management of landslide-prone terrain in the Pacific Northwest. Res. Br., B.C. Min. For., Victoria, B.C. Land Manage. Handb. No. 18.

Chen, J.T. 1991. Edge Effects: Microclimatic Pattern and Biological Responses in Old-growth Douglas-fir Forests. Ph.D. Dissertation, University of Washington, Seattle, Washington.

Christner, J. and R. Harr. 1982. Peak streamflows from the transient snow zone, Western Cascades, Oregon. Pages 27-38 In Proceedings, 50th Western snow conference, Colorado State University press, Fort Collins, Colorado.

Coffin, B.A., and R.D. Harr. 1992. Effects of forest cover on volume of water delivery to soil during rain-on-snow. Final report to Sediment, Hydrology and Mass Wasting Steering Committee, Timber-Fish-Wildlife Group, Project 18.

Collins, B.D. and T. Dunne. 1989. Gravel transport, gravel harvesting, and channel-bed degradation in rivers draining the Southern Olympic Mountains, Washington, USA, Environmental Geology and Water Science, 13, 213-224.

Diller, L.V. and R.L. Wallace. 1996. Distribution and habitat of *Rhyacotriton variegatus* in managed, young growth forests in north coastal California. J. Herpetol. 30:184-191.

Diller, L.V. and R.L. Wallace. 1999. Distribution and habitat of *Ascaphus truei* in streams on managed, young growth forests in north coastal California. J. Herpetol. 33:71-79.

Division of Mines and Geology. 1997. Factors Affecting Landslides in Forested Terrain: California Division of Mines and Geology Note 50, 5 p.

Dryness, C.T. 1969. Hydrologic properties of soils on three small watersheds in the Western Cascades of Oregon. U.S. Forest Ser, Res, Note PNW-111.

Duan, J.G. 2001. Discussion of numerical analysis of river channel processes with bank erosion. Journal of Hydraulic Engineering.

Dunne, T. and L. Leopold. 1978. Water in environmental planning. W. H. Freeman, San Francisco, CA. 818 pp

Durgin, P.P., R.R. Johnston, and A.M. Parsons. 1988. Causes of erosion on private timberlands in northern California. In: Critical Sites Erosion Study: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, v. I, p. 50p.

Everest, F.H., R.L. Beschta, J.C. Scrivner, K,V Koski, J.R. Sedell, and C.J. Cederholm, 1987. Fine sediment and salmonid production: a paradox. Pages 98-142 in Streamside management: forestry and fishery Interactions. E.O. Salo and T.W. Cundy editors. Contribution No. 57. College of Forest Resources, University of Washington, Seattle, Washington.

Furniss, M.J., T.D. Roelofs and C.S. Yee. 1991. Road construction and maintanence. Pages 297-323 in Influences of forest and rangeland management on salmonid fishes and their habitat. W.R. Meehan, editor. Am. Fish. Soc. Spec. Pub. 19.

Furniss, M.J., S.A. Flanagan, and B. McFadin. 2000. Hydrologically-Connected Roads: An Indicator of the Influence of Roads on Chronic Sedimentation, Surface Water Hydrology,and Exposure to Toxic Chemicals. In Stream Notes: To aid in securing favorable conditions of water flows. USDA, Rocky Mountain Research Station. July 2000.

Gibbons, D.R. and E.0. Salo. 1973. An annotated bibliography of the effects of logging on fish of the western United States and Canada. U.S.F.S. Gen.Tech, Rep. PNW-10.

Grant, G.E., F.J. Swanson, and M.G Wolman. 1990. Pattern and origin of stepped-bed morphology in high-gradient streams, western Cascades, Oregon. Geological Society of America Bulletin, 102: 340-352.

Gray, D.H. 1977. Creep Movement and Soil Moisture Stress in Forested vs. Cutover Slopes: Results of Field Studies. University of Michigan, College of Engineering, Department of Civil Engineering, DRDA Project 012577, August 1977.

Gregory, R.S. 1993. Effect of turbidity on the predator avoidance behavior of juvenile chinook salmon (Oncorhynchus tshawytscha). Can. J. Fish. Aquat. Sci. 50:241-246.

Grette, G,B. 1985. The role of large organic debris in juvenile salmonid rearing habitat in small streams. M.S. Thesis, University of Wasfngton, Seattle, Washington.

Hagans, D.K. and W.E. Weaver. 1987. Magnitude, cause and basin response to fluvial erosion, Redwood Creek basin, northern California. 419-428 in Erosion and sedimentation in the Pacific rim. Beschta, R.L., Blinn, T., Grant, G.E., Ice, G.G. and Swanson, F.J. [Eds]. IAHS 165. Washington, DC: International Association of Hydrologic Sciences.Hallock, R.I., R.F, Elwell, and D.H. Fry, Jr. 1970. Migrations of adult king salmon *Oncorhynchus tshawytscha* in the San Joaquin Delta as demonstrated by the use of sonic tags. Calif. Dept. Fish and Came, Fish Bull. 151.

Hallock, R.I., R.F. Elwell, and D.H. Fry, Jr. 1970. Migrations of adult king salmon *Oncorhynchus tshawytscha* in the San Joaquin Delta as demonstrated by the use of sonic tags. Calif. Dept. Fish and Came, Fish Bull. 151.

Harr, R.D. 1977. Water flux in soil and subsoil on a steep forested slope. Journal of Hydrology 33: 37-58.

Harr, R.D, 1986. Effects of clearcurting on rain-on-snow runoff in western Oregon. Water Resources Bulletin 19: 383-393.

Harr, R.D., and F.M. McCorison. 1979. Initial effects of clearcut logging, on size and timing of peak flows in a small watershed in western Oregon. Water Resources Research 15: 90-94.

Harr, R.D., P.L. Fredriksen and J. Rothacher. 1979. Changes in streamflow following timber harvest in southwestern Oregon, U.S.F.S. Res. Paper PNW-249.

Harr, R.D., W.C. Harper, J.T. Krygier and F.S. Hsieh. 1975. Changes in storm hydrographs after road building and clear-cutting in the Oregon Coast Range. Water Resources Research 11: 436-444.

Hartman, G., J.C. Scrivener, L.B. Holtby, and L. Powell. 1987. Some effects of different streamside treatments on physical conditions and fish population processes in Carnation Creek, a coastal rain forest stream in British Columbia. Pages 330-372 in Streamside management, forestry and fishery interactions. E.O. Salo and T.W. Cundy, editors. Contribution No, 57, College of Forest Resources, University of Washington, Seattle, Washington.

Haupt, H.F. 1959. Road and slope characteristics affecting sediment movement from logging roads. Journal of Forestry 57: 329-332.

Hawkins C.P., M.L. Murphy and N.H. Anderson. 1982, Effects of canopy, substrate composition and gradient on the structure of macroinvertebrate communities in Cascade Range streams of Oregon. Ecology *63: 1840*1856.

Hibbert, A.R. 1967. Forest treatment effects on water yield. Pages 527-543 *in* Forest hydrology proceedings, W.E. Sopper and H.W. Lull editors. Pergamon Press, New York.

Hicks, B.J., J.D. Hall, P.A. Bisson and J.R. Sedell. 1991. Responses of salmonids to habitat changes. Pages 483-518 In Influences of forest and rangeland management on salmonid fishes and their habitats. W.R. Meehan, editor. Am. Fish. Soc. Spec. Pub. 19.

Holtby, L.B. 1988. Effects of logging on stream temperatures in Carnation Creek, British Columbia, and associated impacts on the coho salmon (*Oncorhynchus kisutch*). Can. J. Fish. Aq. Sci. 45: 502-51 S.

House, R., V. Crispin, and R. Monthey. 1989. Evaluation of stream restoration projects, Salem District (1981-1988). BLM Tech. Note T/N OR-6.

Hughes, R.M. and G.E. Davis. 1986. Production of coexisting juvenile coho salmon and steelhead trout in heated model stream communities. ASTM Spec. Tech, Pub. 920: 322-337.

Iseya, F. and H. Ikeda. 1987. Pulsations in bedload transport rates induced by a longitudinal sediment sorting: A flume study using sand and gravel mixtures, Geografiska Annaler, 69A, 15-27.

Iverson, R.M. 2000. Landslide triggering by rain infiltration, Water Resources Research v. 36, n. 7, p. 1897-1910.

Johnson, M.G. and R.L. Beschta. 1980. Logging, infiltration capacity, and surface erodibility in western Oregon. J. Forestry. 334-337.

Kelsey, H.M. 1978, Earthflows in Franciscan melange, Van Duzen River basin, California. Geology 6: 361-364.

Kelsey, H.M. 1980. A sediment budget and an analysis of geomorphic process in the Van Duzen River basin, north coastal California, 1941-1975. Geol. Soc. Am. Bulletin 91:1119-1216.

Kelsey, H.M. 1982. Hillslope evolution and sediment movement in a forested headwater basin, Van Duzen River, north coastal California. PNW-141, Pacific Northwest Forest and Range Experiment Station.

Keppeler, E.T. 1998. The summer flow and water yield response to timber harvest. In: Ziemer, Robert R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story, 6 May 1998; Ukiah, California. General Tech. Rep. PSW GTR-168. Albany, California: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 35-43.

Keppeler, E., and D. Brown, 1998. Subsurface Drainage Processes and Management Impacts.USDA Forest Service Gen. Tech. Rep. PSW-GTR-168. 1998.

Keppeler, E.T., and R.R. Ziemer. 1990. Logging effects on streamflow: water yields and summer flows at Caspar Creek in northwestern California. Water Resources Research 26(7): 1669-1679.

Keppeler, E.T., R.R. Ziemer, and P.H. Cafferata. 1994. Changes in soil moisture and pore pressure after harvesting a forested hillslope in northern California. Pages 205-214, in: Marston, Richard A., and Victor R. Hasfurther (eds). Proceedings, Annual Summer Symposium of the American Water Resources Association: Effects of Human-Induced Changes on Hydrologic Systems, 26-29 June 1994, Jackson Hole, Wyoming. American Water Resources Association, Bethesda, Maryland.

Keppeler, E., and D. Brown. 1998. Subsurface Drainage Processes and Management Impacts.USDA Forest Service Gen. Tech. Rep. PSW-GTR-168. 1998.

King, J.C., and L.C. Tennyson. 1984. Alteration of streamflow characteristics following road construction in north central Idaho. Water Resources Research 20; 1159-1163.

Krogstad, F. 1995. A physiology and ecology based model of lateral root reinforcement of unstable hillslopes, MS thesis, University of Washington, Seattle, WA, 44 pages.

Larse, R.W. 1971. Prevention and control of erosion and stream sedimentation from forest roads. Pages 76-83 in Proceedings of forest land uses and stream environment symposium. J.T. Krygier and J.D. Hall editors. Cont. Ed, Publ. Oregon State University, Corvalis, Oregon.

Ledwith, T. 1996. The effects of buffer strip width on air temperature and relative humidity in a stream riparian zone. Watershed Management Council Networker, Summer.

Lehre, A.K., and G. Carver. 1985. Thrust faulting and earthflows: speculations on the sediment budget of a tectonically active drainage basin, in M.E. Savina, ed.: American Geomorphological Field Group Field Trip Guidebook 1985 Conference, p. 169-183.

Lewis, J. 1998. Evaluating the impacts of logging activities on erosion and sediment transport in the Caspar Creek watersheds. In: Ziemer, Robert R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story, 6 May 1998; Ukiah, California. General Tech. Rep. PSW GTR-168. Albany, California: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 55-69.

Lisle, T.E. 1990. The Eel River, northwestern California; high sediment yields from a dynamic landscape. Pages 311-314, in: M.G. Wolman and H.C. Riggs (ed.), Surface Water Hydrology, v. O-1, The Geology of North America, Geological Society of America.

Lisle, T.E. and S. Hilton. 1999. Fine bed material in pools of natural gravel bed channels. Water Resources Research 35(4): 1291-1304

Lisle, T.E., J.E. Pizzuto, H. Ikeda, F. Iseya, and Y. Kodama. 1997. Evolution of a sediment wave in an experimental channel. Water Resources Research 33(8): 1971-1981.

Lisle, T.E. 1989. Sediment transport and resulting deposition in spawning gravels, North Coastal California. Water Resources Research 25: 1303-1319.

Lisle, T.E., J.M. Nelson, J. Pitlick, M.A. Madej, and B.L. Barkett. 2000. Variability of bed mobility in natural, gravel-bed channels and adjustments to sediment load at local and reach scales. Water Resources Research, 36(12): 3743-3755.

Lisle, T.E., Y. Cui, G. Parker, J.E. Pizzuto, and A.M. Dodd. (in press). The dominance of dispersion in the evolution of bed material waves in gravel-bed rivers. Earth Surface Processes and Landforms.

Luce, C.H., and T.A. Black. 1999. Sediment production from forest roads in western Oregon. Water Resources Research. 35(8):2561-2570.

Marron, D.C., K.M. Nolan, and R.J. Janda. 1995. Surface erosion by overland flow in the Redwood Creek basin, northwestern California, effects of logging and rock type. 1454, USGS.

Maser,C. and J.R. Sedell. 1994. From the forest to the sea: the ecology of wood in streams, rivers, estuaries, and oceans. St. Lucie Press, Delray Beach, Florida. 200 pp.

McCashion, J.D., and R.M. Rice. 1983. Erosion on logging roads in northwestern California: How much is avoidable?: Journal of Forestry, v. 81, p. 23-26.

Metter, D.E. 1964. A morphological and ecological comparison of two populations of the tailed frog, *Ascaphus truei* Stejneger. Copeia 1964:181-204.

Miller, D.J. and J. Sias. 1998. Deciphering large landslides: linking hydrological, groundwater and slope stability models through GIS. Hydrological Processes 12:923-941.

Montgomery, D.R. and J.M. Buffington. 1993. Channel classification, prediction of channel response, and assessment of channel condition. Report TFW-SH10-93-002 prepared for the SHAMW committee of the Washington State Timber/Fish/Wildlife Agreement, 84 pgs.

Montgomery, D.R., K.M. Schmidt, H. Greenberg and W.E. Dietrich. 2000. Forest clearing and regional landsliding, Geology, v. 28, p. 311-314.

Murphy M.L. and W.R. Meehan. 1991. Stream Ecosystems. American Fisheries Society Special Publication 19:139-179.

Murphy, M.L., J. Heifetz, S.W. Johnson, K.V. Koski and J.F. Thedinga. 1986. Effects of clear-cut logging with and without buffer strips on juvenile salmonids in Alaskan streams. Canadian Journal of Fisheries and Aquatic Sciences, 43: 1521-151533.

NCASI (National Council of the Paper Industry for Air and Stream Improvement, Inc.). 1999. Scale considerations and the detectability of sedimentary cumulative watershed effects. Technical Bulletin No. 776. Research Triangle Park, N.C., National Council of the Paper Industry for Air and Stream Improvement, Inc.

Newcombe, C.P. and D.D. MacDonald. 1991. Effects of suspended sediment on aquatic ecosystems. N. Amer. J. Fish. Man. 11: 72-82.

Newcombe, C.P. and J.O.T. Jensen. 1996. Channel suspended sediment and fisheries: A synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16: 693-727.

Nolan, K.M. and R.J. Janda. 1995a. Movement and sediment yield of two earthflows, northwestern California IN Nolan, K.M. Kelsey, H.M., and D.C. Marron (eds.) Geomorphic Processes and Aquatic Habitat in the Redwood Creek Basin, Northwestern California, U.S. Geological Survey Professional Paper 1454, U.S. Government Printing Office, 1995.

Nolan, K.M. and R.J. Janda. 1995b. Impacts of logging on stream-sediment discharge in the Redwood Creek basin, northwestern California IN Nolan, K.M. Kelsey, H.M., and D.C. Marron (eds.) Geomorphic Processes and Aquatic Habitat in the Redwood Creek Basin, Northwestern California, U.S. Geological Survey Professional Paper 1454, U.S. Government Printing Office, 1995.

O'Connor, M.D. and R.D. Harr. 1994. Bedload transport and large organic debris in steep mountain streams in forested watersheds on the Olympic Peninsula, Washington. Final Report to Washington Department of Natural Resources and Timber/Fish/Wildlife, Olympia, Washington. TFW-SH7-94-001.

O'Loughlin, C., and R.R. Ziemer. 1982. The importance of root strength and deterioration rates upon edaphic stability in steepland forests. Proceedings of I.U.F.R.O. Workshop P.1.07-00 Ecology of Subalpine Ecosystems as a Key to Management. 2-3 August 1982, Corvallis, Oregon. Oregon State University, Corvallis, Oregon. pp. 70-78.

Poulin, V.A. 1984. A research approach to solving fish-forestry interactions In the Queen Charlotte Islands. B.C. Ministry of Forests, Land Man. Rep. 27. Victoria, British Columbia.

PWA. 1998a. Sediment source investigation and sediment reduction plan for the North Fork Elk River watershed, Humboldt County, CA., Unpublished technical report for Pacific Lumber Company.

PWA. 1998b. Sediment Source Investigation and Sediment Routing Plan for the Bear Creek Watershed, Humboldt County, Pacific Watershed Associates, Technical report prepared for The Pacific Lumber Company, Scotia, CA, April, 1998, 42 p., p. 42.

PWA. 1999a. Sediment Source Investigation and Sediment Reduction Plan for Freshwater Creek Watershed, Humboldt County, CA, Technical report prepared for Pacific Lumber Company, Scotia, CA, by Pacific Watershed Associates, p. 94.

PWA. 1999b. Sediment Source Investigation and Sediment Reduction Plan for the Jordan Creek Watershed, Humboldt County, CA, Pacific Watershed Associates, Technical report prepared for The Pacific Lumber Company, Scotia, CA, January, 1999, 63p p.

Pyles, M.R. K. Mills and G. Saunders. 1987. Mechanics and stability of the Lookout Creek earth flow. Bulletin of the Association of Engineering Geologists. 24(2): 267-280.

Raines, M.A., and H.M. Kelsey. 1991. Sediment budget for the Grouse Creek basin, Humboldt County, CA., USDA, USDA Forest Service, Six Rivers National Forest, Eureka, CA.

Raudkivi, A.J. 1990, Loose Boundary Hydraulics, Pergamon Press, Elmsford, NYReeves, G.H., F.H. Everest, and J.D. Hall. 1987. Interactions between the redside shiner *(Richardsonius balteatus)* and the steelhead trout *(Salmo gairdneri)* in western Oregon: the influence of water temperature, Can. J. Fish. Aq. Sci. 44: 1603-1613.

Reeves, G.H., F.H. Everest, and J.D. Hall. 1987. Interactions between the redside shiner (Richardsonius balteatus) and the steelhead trout (Salmo gairdneri) in western Oregon: the influence of water temperature, Can. J. Fish. Aq. Sci. 44: 1603-1613.

Reid, L.M. and T. Dunne. 1984. Sediment Production from forest road surfaces. Water Resources Research 20; 1753-1761.

Reid, L.M., and T. Dunne. 1996. Rapid Evaluation of Sediment Budgets. Catena Verlag GMBH, Reiskirchen, Germany. 164 p.

Reiser, D.W. and R.G. White. 1988. Effects of two sediment-size classes on steelhead trout and Chinook salmon egg incubation and juvenile quality. North American Journal of fisheries Management 8:432-437.

Rice, R.M., and P.A. Datzman. 1981. Erosion associated with cable and tractor logging in northwestern California. In: Timothy R. H. Davies and Andrew J. Pearce (eds.), Erosion and Sediment Transport in Pacific Rim Steeplands, Proceedings of the Christchurch Symposium, 25-31 January 1981, Christchurch, New Zealand. Int. Assn. Hydrol. Sci. Pub. No. 132: 362-374.

Rice, R.M., and J. Lewis. 1991. Estimating erosion risks associated with logging and forested roads in northwestern California.: Water Resources Bulletin, v. 27, p. 809-818.

Richards, K.S. 1982. Rivers: form and process in alluvial channels. Methuen. London.

Robison, E.G., K. Mills, J. Paul, L. Dent, and A. Skaugset. 1999. Storm Impacts and Landslides of 1996: Final Report; Forest Practices Technical Report Number 4, Oregon Department of Forestry, p. 144.

Rood, K.M. 1984. An aerial photograph inventory of the frequency and yield of mass wasting an the Queen Charlotte Islands, British Columbia. B.C. Ministry of Forests and Land Man. Rep. 34, Victoria, British Columbia.

Schmidt, K.M. Roering, J.J., Stock, J.D., Dietrich, W.E., Montgomery, D.R., and Schaub, T. (in press) Root cohesion variability and susceptibility to shallow landsliding in the Oregon Coast Range. Canadian Geotechnical Journal.

Sedell, J.R. and J.L. Froggart. 1984. Importance of streamside forests to large rivers: the isolation of the Willamette River, Oregon, USA from its floodplain by snagging and streamside forest removal. Int. Vere. Lim. Verhandlungen, 22: 1828-1834.

Sedell, J.R., F.N. Leone, and W.S. Duval. 1991. Water transportation and storage of logs. Pages 325-368 in Influences of forest and rangeland management on salmonid fishes and their habitats. W.E. Meehan, editor. A.F.S. Spec. Pub. 19, Bethesada, Maryland.

Sidle, R.C. and D.N. Swanston. 1982. Analysis of a small debris slide in coastal Alaska. Can. Geotech. J. 19(2): 167-174.

Sidle, R.C. 1992. A theoretical model of the effects of timber harvesting on slope stability. Water Resour. Res. 28(7): 1897-1910.

Sidle, R.C., A.J. Pearce and C.L. O'Loughlin. 1985. Hillslope stability and land use. Water Resources Monograph Series 11.

Sigler, J. W., T. C. Bjornn, and F. H. Everest. 1984. Effects of chronic turbidity on density and growth of steelheads and coho salmon. Transactions of the American Fisheries Society 113:142-150.

Sowma-Bawcom, J.A. 1996. Breached landslide dam on the Navarro River. California Geology, 49(5): 120-127

Spence, B.C., G.A. Lomnicky, R.M. Hughes, R.P. Novitzki. 1996. An ecosystem approach to salmonid conservation. TR-4501-96-6057. Corvallis, OR. Man Tech Environmental Research Services Corporation.

Steele, J. and G. Stacey. 1994. Coho salmon habitat impacts: a qualitative assessment technique for registered professional foresters. CDFG, Draft #2. 31 pp.

Sullivan, K., T.E. Lisle, C.A. Dolloff, G.E. Grant, and L.M. Reid. 1987. Stream channels; the link between forests and fishes. Pages 39-97 in Salo and Cundy (1987).

Sullivan, K., J. Tooley, K. Doughty, J.E. Caldwell, and P. Knudsen. 1990. Evaluation of prediction models and characterization of stream temperature regimes in Washington. Timber/Fish/Wildlife Rep. No. TFW-WQ3-90-006. Washington Dept. Nat. Resources, Olympia, Washington. 224 pp.

Swanson, F.J. 1981. Fire and geomorphic processes. Pp. 410- 420. In: Mooney, H. et al. (eds.) Proceedings of the Conference on Fire Regimes and Ecosystem Properties, Gen. Tech. Rep. WO-26. Washington, DC: US Department of Agriculture, Forest Service.

Swanson, F.J., T.K. Kratz, N. Caine, and R.G. Woodmansee. 1988. Landform effects on ecosystem patterns and processes. Bioscience 38: 92-98.

Swanson, F.J., and C.T Dryness. 1975, Impact of clearcutting and road construction on soil erosion by landslides in the western Cascade Range, Oregon.: Geology, v. 3, p. 393-396.

Swanson, F.J., L.E. Benda, S.H. Duncan, G.E. Grant, W.F. Megahan, L.M. Reid and R.R. Zeimer. 1987. Mass failures and other processes of sediment production in Pacific Northwest forest landscapes. Pages 9-38 In Streamside management: forestry and fishery interactions. E.O. Salo and T.W. Cundy editors. Contribution No. 57. College of Forest Resources, University of Washington, Seattle, Washington.

Swanston, D.N. 1981. Creep and earthflow erosion from undisturbed and management impacted slopes of the Coast and Cascade Ranges of the Pacific Northwest, U.S.A. In: Erosion and Sediment Transport in Pacific Rim Steeplands. I.A.H.S. Publ. No. 132:76-94.

Swanston, D.N. and F.J. Swanson, 1976. Timber harvesting, mass erosion, and steepland forest geomorphology in the Pacific Northwest. In: Coates, Donald R., ed. Geomorphology and engineering. Stroudsburg, PA: Dowden, Hutchinson & Ross, Inc.: 199-221.

Swanston, D.N. 1971. Principal mass movement processes influenced by roadbuilding, logging and fire. Pages 29-40 In Proceedings of forest land uses and stream management symposium, Cont. Ed. Publ. Oregon State University, Corvalis, Oregon.

Swanston, D.N. 1991. Natural processes. Pp. 139-179 In: W.R. Meehan (editor), Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. Am. Fish. Soc. Special Publication No. 19.

Transportation Research Board. 1996. Landslides: Investigation and Mitigation, Special Report 247. Turner, A.K. and Schuster, R.L. (Eds.) National Research Council, National Academy Press, Washington D.C., 673 p.

Triska, F.J., V.C. Kennedy, R.J. Avanzino and B.N. Reilly. 1983. Effect of simulated canopy cover on regulation of nitrate uptake and primary production by natural periphyton assemblages. pp 129-159 In: T. Fontaine and S. Bartell (eds.), Dynamics of L otic Ecosystems, Ann Arbor Science Publishers, Michigan.

Watershed Professionals Network, 2001. Freshwater Creek Watershed Analysis. Prepared for Pacific Lumber Company (Palco) Scotia, CA. Jan. 2001.

Weaver, W.E. and D.K. Hagans, 1994. Handbook for forest and ranch roads; a guide for planning, designing, constructing, reconstructing, maintaining and closing wildland roads. Pacific Watershed Associates, Arcata, California. 190 pp.

Welch. E., J.M. Jacoby and C.W. May. 1998. Stream quality. In R. J. Naiman 8r R. E. Bilhy teds). River Ecology and Management: Lessons from the Paatic Coastal Ecoregion. Springer-Verlag. New York: 67-92.

Welsh, H.H., Jr., and A.J. Lind. 1996. Habitat correlates of the southern torrent salamander, Rhyacotriton variegates (Caudata: Rhyacotritonidae) in northwestern California. Journal of Herpetology 30:385-398.

Wieczorek, G.F., 1996, Landslide triggering mechanisms; in, Landslides Investigation and Mitigation, A. K. Turner and R. L. Schuster, eds.: National Research Council, Transportation Research Board, Special Report 247, p. 76-90.

Wood, P.J. and P.D Armitage. 1997. Biological effects of fine sediment in the lotic environment Environmental Management, 21, 203-217.

Wright, K.A., K.H. Sendek, R.M. Rice, and R.S. Thomas. 1990. Logging effects on streamflow: storm runoff at Caspar Creek in northwestern California. Water Resources Research, 26(7)- 1657-1667.

Wu, T. H. and D.N. Swanston, 1980. Risk of landslide in shallow soils and its relation to clearcutting in southeastern Alaska, For. Sci., 26(3), 495-510.

Yang, C.T. 1996. Sediment Transport: Theory and Practice. McGraw Hill, 396 p.

Yoshinori, T. and K. Osamu. 1984. Vegetative influences on debris slide occurrences on steep slopes in Japan In Symposium on effects of forest land use on erosion and slope stability. East-West Center, Honolulu. pp. 63-72.

Ziemer, R.R. 1981a. Roots and the stability of forested slopes. In: Timothy R. H. Davies and Andrew J. Pearce (eds.), Erosion and Sediment Transport in Pacific Rim Steeplands, Proceedings of the Christchurch Symposium, 25-31 January 1981, Christchurch, New Zealand. Int. Assn. Hydrol. Sci. Pub. No. 132: 343-361.

Ziemer, R.R., and D.N. Swanston. 1977. Root strength changes after logging in southeast Alaska. U.S. Dept. Agric., Forest Service, Research Note PNW-306, Portland, Oregon. 10 pp.

Ziemer, R.R. 1981b. Some effects of silvicultural options on the stability of slopes. In: Research on the Effects of Mass Wasting of Forest Lands on Water Quality and the Impact of Sediment on Aquatic Organisms, Stream Improvement Tech. Bull. No. 344. National Council of the Paper Industry for Air and Stream Improvement, New York, N.Y. 6-17.

Ziemer, R.R. 1981c. The role of vegetation in the stability of forested slopes. Proceedings of the International Union of Forestry Research Organizations, XVII World Congress, 6-17 September 198l, Kyoto, Japan. vol. I: 297-308.

Ziemer, R.R. 1998. Flooding and stormflows. In: Ziemer, Robert R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story, 6 May 1998; Ukiah, California. General Tech. Rep. PSW GTR-168. Albany, California: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 15-24.